U.S. DEPARTMENT OF COMMERCE
National Technical Information Service
PB-299 657

Applicability of Land Treatment of Wastewater in the Great Lakes Area Basin: Impact of Wastewater Diversion, Spray Irrigation on Water Quality in the Muskegon County, Michigan Lakes

Michigan Univ., East Lansing

Prepared for

Environmental Protection Agency, Chicago, IL. Great Lakes National Program Office

May 79

United States
Environmental Protection
Agency

Great Lakes National
Programs Office
Room 932, 536 S. Clark St.
Chicago, Illinois 60605

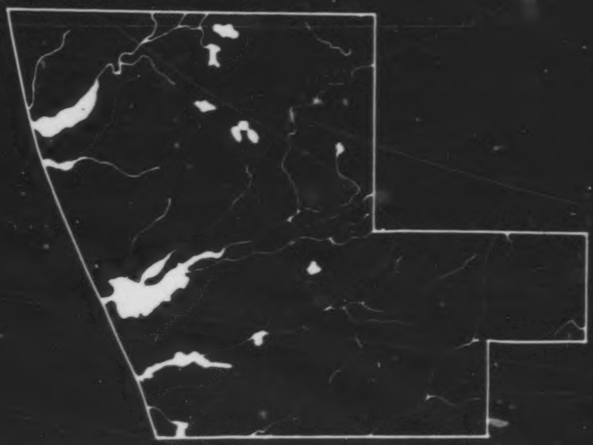
EPA-905/9-79-006-A

PB 299657

EPA

Applicability of Land Treatment of Wastewater in the Great Lakes Area Basin

Impact of Wastewater Diversion, Spray Irrigation on Water Quality in the Muskegon County, Michigan Lakes



REPRODUCED BY
NATIONAL TECHNICAL
INFORMATION SERVICE
U. S. DEPARTMENT OF COMMERCE
SPRINGFIELD, VA. 22161

MCD-56

Preface

The U. S. Environmental Protection Agency was created because of increasing public and governmental concern about the dangers of pollution to the health and welfare of the American people. Noxious air, foul water, and spoiled land are tragic testimony to the deterioration of our natural environment.

The Great Lakes National Program Office (GLNPO) of the U.S. EPA was established in Region V, Chicago, to provide specific focus on the water quality concerns of the Great Lakes. The Section 108(a) Demonstration Grant Program of the Clean Water Act (PL92-500) is specific to the Great Lakes drainage basin and thus is administered by the Great Lakes National Program Office.

Land disposal of wastewater in the Great Lakes area is one alternative for treatment that can provide tertiary quality effluent when properly managed. This report evaluates the impact of wastewater diversion and spray irrigation on water quality in the Muskegon County, Michigan lakes.

We hope the information and data contained herein will help planners and managers of pollution control agencies to make better decisions in carrying forward their pollution control responsibilities.

Edith J. Tebo, Ph.D.
Director
Great Lakes National Program Office

TECHNICAL REPORT DATA
(Please read Instructions on the reverse before completing)

1. REPORT NO.
EPA-905/9-79-006-A

2.

3. RECIPIENT'S ACCESSION NO.
PB299657

4. TITLE AND SUBTITLE
Applicability of Land Treatment of Wastewater in The Great Lakes Area Basin - Impact of Wastewater Diversion Spray Irrigation on Water Quality in Muskegon County, Michigan Lakes

5. REPORT DATE
May 1979

6. PERFORMING ORGANIZATION CODE

7. AUTHOR(S)
P.L. Freedman, R.P. Canale, and M.T. Auer

8. PERFORMING ORGANIZATION REPORT NO.

9. PERFORMING ORGANIZATION NAME AND ADDRESS
Department of Civil Engineering
The University of Michigan
Ann Arbor, Michigan 48105

10. PROGRAM ELEMENT NO.
2BA645

11. CONTRACT/GRANT NO.
G005104 01

12. SPONSORING AGENCY NAME AND ADDRESS
U.S. Environmental Protection Agency
Great Lakes National Program Office
536 South Clark Street, Room 932
Chicago, Illinois 60605

13. TYPE OF REPORT AND PERIOD COVERED
Final-April 1972-June 1976

14. SPONSORING AGENCY CODE

15. SUPPLEMENTARY NOTES
EPA Project Officers: Dr. John M. Walker and Stephen Poloncsik

16. ABSTRACT
The Muskegon County Wastewater Management System is a lagoon impoundment, spray irrigation facility which treats about 102,000 cubic meters of wastewater per day and irrigates 2,160 hectares of corn land. Irrigated treated water channels back to the surface water via subsurface drains, percolation through the soil, wells and open ditches. A significant amount of the wastewater was diverted to the treatment site in 1974 from out-dated inefficient treatment plants which had previously discharged poorly treated wastewater directly to the County's surface waters.

Data was collected from 1972 to 1975 to determine the impact of the diversion of wastewater and subsequent treatment as the site on improved quality of surface waters. Within this.study, particular emphasis was placed on an analysis of the effect of the diversion project on the trophic status and associated nutrient budgets of three major County lakes. Organics, trace metals and suspended solids were not measured. In this effort, tributary data was combined with information on municipal and industrial nutrient loads.

The Data shows improvement of water quality in two of the three lakes where non-point sources of pollution were not dominant. Eutrophication models show that the current reduction in nutrient loadings will not be sufficient to change the eutrophic status of the lakes. However, additional improvement in water transparency, oxygen conditions and other water quality parameters are expected.

17. KEY WORDS AND DOCUMENT ANALYSIS

a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group
Water Quality Land Use Land treatment of Wastewater Lake Renovation Eutrophic Muskegon		

18. DISTRIBUTION STATEMENT Available to Municipal Construction Div. Denver, CO, Great Lakes National Program Office Chgo., IL and NTIS, Springfield, VA 22151

19. SECURITY CLASS (This Report)

20. SECURITY CLASS (This page)

21. NO. OF PAGES
278

22. PRICE
A13-A01

EPA Form 2220-1 (Rev. 4-77) PREVIOUS EDITION IS OBSOLETE

☆ U.S. GOVERNMENT PRINTING OFFICE 1979—652-042

EPA-905/9-79-006-A
May 1979

APPLICABILITY OF LAND TREATMENT OF WASTEWATER IN THE GREAT LAKES AREA BASIN:

IMPACT OF WASTEWATER DIVERSION, SPRAY IRRIGATION ON WATER QUALITY IN THE MUSKEGON COUNTY, MICHIGAN, LAKES

by

P. L. Freedman, R. P. Canale, and M. T. Auer

Department of Civil Engineering
The University of Michigan
Ann Arbor, Michigan 48105

for

Michigan Water Resources Commission
Department of Natural Resources
Lansing, Michigan 48926

EPA Grant No. G005104

Project Officers
J. M. Walker and S. Poloncsik
Office of Research and Development

SECTION 108 (a) PROGRAM
GREAT LAKES NATIONAL PROGRAM OFFICE
U.S. ENVIRONMENTAL PROTECTION AGENCY, REGION V
CHICAGO, ILLINOIS 60605

i.a

FOREWORD

The Great Lakes are the world's largest fresh water resource. It is shared by the United States and Canada. In the U.S., this lake system serves important water needs of people and industries in eight states, six of which comprise the area served by EPA Region V. The usefulness of the lakes, for many purposes, has been impaired by past misuses of the lakes themselves and of land resources in contributing drainage basins. Management of the Great Lakes Area Basin to halt or reverse the degradation of vital water resources is of great importance to both countries. Lake Michigan presents special concerns because of its headwater relationship to the lower Great Lakes and because of the intensity and variety of human activity that impacts upon it. Responsibilities for developing and enforcing ameliorative management in the Lake Michigan basin rest with the U.S. and the four states that share the shoreline of the lake.

Land application of wastewater is one of the management options for upgrading water after use. From historical precedent and for many theoretical reasons, land application has the potential for effecting full renovation of wastewater before release into the environment. Whether this potential is realized will depend on many factors of soil, climate and management which must be understood for each situation. Performance must be assessed ultimately in terms of impact on contiguous aquatic systems. The acquisition of background and early operational data for a large land application system in Muskegon County, Michigan, has been the objective of an intensive three-year study conducted for EPA Region V by the Michigan Water Resources Commission, with subcontracts to Michigan State University and the University of Michigan. The three reports covering this work carry the general title, "Applicability of Land Treatment of Wastewater in the Great Lakes Area Basin," with respective subtitles:

> The Muskegon County System--An Overview, Monitoring Considerations and Impacts on Receiving Waters.
>
> Effectiveness of Sandy Soils at Muskegon County, Michigan, for Renovating Wastewater.
>
> Impact of Wastewater Diversion, Spray Irrigation on Water Quality in Muskegon County, Michigan, Lakes.

In these volumes, data collected from 1972 through 1975 are evaluated in relation to the applicability of land treatment for renovating municipal and industrial wastewaters in Muskegon County. Short-term and long-term projections are made regarding management practices that can influence the renovative effectiveness of soils and crops. Observed and projected effects of wastewater diversion and treatment on water quality and ecosystem responses in lakes and streams that drain into Lake Michigan are described.

EPA REVIEW NOTICE

This report has been reviewed by the Office of Research and Development and the Great Lakes National Program Office of Region V, U.S. EPA, Chicago, and approved for publication. Approval does not signify that the contents necessarily reflect the views and policies of the Environmental Protection Agency, nor does mention of trade names or commercial products constitute endorsement or recommendation for use.

ACKNOWLEDGMENTS

This project was carried out under the auspices of The University of Michigan's Great Lakes Resource Management Program (GLRMP). Dr. John Armstrong is Director of GLRMP and was largely responsible for the conception and initial design of the project. He provided many important ideas concerning the formulation and implementation of the overall research plan.

Dr. Peter Meier and Thomas Kelly assisted with the design and implementation of the field monitoring program. The authors appreciate the help of Bruce Bartley, Michael Gould, and Paul Silfven for their efforts with sample collection and field water quality analysis. The majority of the nutrient analysis was conducted by Mary Lee Sharp under the direction of Dr. K. H. Mancy.

Dr. John Walker (US Environmental Protection Agency Project Coordinator) provided several valuable suggestions concerning the presentation of the results and compiled the data presented in Table 1 and Appendix E.

Warren Slocum (State of Michigan Environmental Protection Bureau) helped summarize the industrial discharge data and inventory presented in Appendix F.

Various sections and drafts of the manuscript were typed by Lee Hallmark, Diane Rumps, and Aleda Thomas. Mrs. Thomas was also responsible for the final editing of the report and without her diligence and perseverence the report could not have been produced.

TABLE OF CONTENTS

Page

FOREWORD i
EPA REVIEW NOTICE. ii
ACKNOWLEDGMENTS. iii
LIST OF TABLES viii
LIST OF ILLUSTRATIONS. x

SECTIONS

I. SUMMARY AND CONCLUSIONS. 1
  WHITE LAKE 2
  MUSKEGON LAKE. 3
  MONA LAKE. 4
II. RECOMMENDATIONS. 8
  FUTURE WATER QUALITY STUDIES 8
  MANAGEMENT RECOMMENDATIONS 9
III. INTRODUCTION 11
IV. DESCRIPTION OF STUDY AREA AND SOCIO-ECONOMIC HISTORY 13
V. HISTORY AND OPERATION OF THE MUSKEGON DIVERSION SPRAY-IRRIGATION SYSTEM 17
VI. METHODS AND MATERIALS. 21
  GENERAL 21
    Sampling Methods 28
  PHYSICAL AND CHEMICAL METHODS OF ANALYSIS 29
    Alkalinity 29
    Biochemical Oxygen Demand. 29
    Chemical Oxygen Demand 32
    Chloride 32
    Dissolved Oxygen 32
    Hydrogen Ion Activity. 32
    Iron 33
    Ammonia Nitrogen 33
    Nitrate, Nitrite Nitrogen. 33
    Phosphorus 33
    Relative Irradiance. 34

SECTIONS Page

Solar Radiation 34
Silicon 34
Specific Conductance 34
Total Organic Carbon 34
Temperature 35
Transparency 35

BIOLOGICAL METHODS OF ANALYSIS 35

Chlorophyll *a* 35
Primary Productivity 35

VII. GENERAL LIMNOLOGICAL OBSERVATIONS 37

INFORMATION 37

PHYSICAL AND CHEMICAL PARAMETERS 37

pH and Alkalinity 37
Conductivity and Chlorides 40
Oxygen and Oxygen Demanding Substance 40
Nitrogen 41
Phosphorus 42
Silicon 42
Iron 43

BIOLOGICAL PARAMETERS

Chlorophyll *a* 43
Primary Productivity 43
Secchi Disc 44
Phytoplankton, Zooplankton, and Benthos 44

SUMMARY 44

VIII. WHITE LAKE 45

INTRODUCTION 45

TRIBUTARY-RELATED CONSIDERATIONS 46

Hydrology 46
Concentration of Chemical Species in Tributaries 50
Nutrient Loads 64
Summary 81

LAKE-RELATED CONSIDERATIONS 81

Spatial and Seasonal Distributions 81
Long-Term Changes 100
Summary 103

SECTIONS Page

SPECIALTY STUDIES. 107

Algal Growth Nutrient Bioassays 107
Macrophytes 110
Sediment-Nutrient Relationships 114
Modeling. 117

IX. MUSKEGON LAKE 124

INTRODUCTION 124

TRIBUTARY-RELATED CONSIDERATIONS 124

Hydrology 124
Concentrations of Chemical Species in Tributaries . . 126
Nutrient Loads. 137
Summary 152

LAKE-RELATED CONSIDERATIONS. 152

Spatial and Seasonal Distributions. 152
Long-Term Changes 169
Summary 171

X. MONA LAKE 175

INTRODUCTION 175

TRIBUTARY-RELATED CONSIDERATIONS 175

Hydrology 175
Concentrations of Chemical Species in Tributaries . . 177
Nutrient Loads. 188
Summary 201

LAKE-RELATED CONSIDERATIONS. 201

Spatial and Seasonal Distributions. 201
Long-Term Changes 215
Summary 220

LITERATURE CITED 222

APPENDICES

A - MUSKEGON ALGAL NUTRIENT BIOASSAY STUDY, WHITE LAKE PROJECT 227

B - SUBMERGED AQUATIC MACROPHYTES IN WHITE LAKE, MICHIGAN 229

APPENDICES Page

C - NUTRIENT RELEASE FROM ANAEROBIC SEDIMENTS IN WHITE LAKE, MICHIGAN. 231

D - MODEL PROJECTIONS OF PHOSPHORUS CONCENTRATIONS. 233

E - WATER QUALITY AND LOADING RATE DATA FOR THE MUSKEGON COUNTY TREATMENT SYSTEM. 236

F - INDUSTRIAL AND MUNICIPAL DISCHARGE INVENTORY. 248

G - AVAILABILITY OF DATA FROM U.S. EPA STORAGE AND RETRIEVAL COMPUTER SYSTEM "STORET" 258

LIST OF TABLES

Table	Page
1. Chronology of some important events relevant to the Muskegon County Wastewater Treatment Project	19
2. Sampling cruse schedule	22
3. Location of tributary sampling stations	23
4. Summary of methods of analysis	30
5. USEPA interlaboratory comparison study	31
6. Water quality characteristics—averages of all data	38
7. White Lake drainage basin areas	47
8. White Lake tributary flow data, July 11 and 12, 1967 (MWRC, 1967)	48
9. Average concentrations of selected chemical species in tributaries to White Lake, 1972-1975	51
10. Estimated municipal and industrial loads to White Lake prior to diversion in kilograms per day	66
11. Annual average White Lake tributary loads in kilograms per day	68
12. Estimated total yearly nutrient loads to White Lake, 1972-1975 in thousand kilograms per year	73
13. Normalized yearly average White River loads in kilograms per day	75
14. Flow weighted average concentrations of total phosphorus and dissolved inorganic nitrogen in tributaries to White Lake	78
15. Key to symbols used in Figure 25 (after Vollenweider, 1975)	80
16. Yearly average flux of nutrients leaving and entering White Lake in kilograms per day	82
17. Average annual values for selected water quality parameters in White Lake, 1972-1975	102
18. Muskegon Lake drainage basin characteristics	125
19. Average concentrations of selected chemical species in tributaries to Muskegon Lake, 1972-1975	128
20. Estimated municipal and industrial loads to Muskegon Lake prior to diversion in kilograms per day	140

Table Page

21. Average Muskegon Lake tributary loads in kilograms per day. . . 141

22. Estimated total yearly nutrient loads to Muskegon Lake, 1972-1975 in thousand kilograms per year. 146

23. Normalized yearly average Muskegon River loads in kilograms per day 148

24. Flow weighted average concentrations of total phosphorus and dissolved inorganic nitrogen in tributaries to Muskegon Lake 151

25. A summary of the yearly average flux of nutrients leaving and entering Muskegon Lake in kilograms per day 153

26. Average annual values for selected water quality parameters in Muskegon Lake, 1972-1975 170

27. Mona Lake drainage basin characteristics (projected long-term average) 176

28. Average concentrations of selected chemical species in tributaries to Mona Lake, 1972-1975 178

29. Average Mona Lake tributary loads in kilograms per day. 190

30. Estimated municipal and industrial loads to Mona Lake prior to diversion in kilograms per day 196

31. Estimated total yearly nutrient loads to Mona Lake, 1972-1975 in thousand kilograms per year. 199

32. Flow weighted average concentrations of total phosphorus and dissolved inorganic nitrogen in tributaries to Mona Lake. 200

33. A summary of the yearly average flux of nutrients leaving and entering Mona Lake in kilograms per day 202

34. Algicide application to Mona Lake (Michigan DNR, 1976). 218

35. Average annual values for selected water quality parameters in Mona Lake, 1972-1975 219

LIST OF ILLUSTRATIONS

Figure Page

1. Annual average nutrient loadings to the Muskegon Lakes. 6

2. Annual average surface water concentrations of selected water quality variables in the Muskegon Lakes 7

3. The Muskegon Lakes study area 14

4. White Lake sampling stations. 24

5. Muskegon Lake sampling stations 25

6. Mona Lake sampling stations 26

7. Tributary and lake sampling stations. 27

8. White River average annual flow at Whitehall, Michigan (USGS Station # 04122200) 49

9. Mean monthly flow of the White River at Whitehall, Michigan (USGS Station # 04122200). 49

10. Dissolved inorganic nitrogen concentrations at Station 201 in White River, 1972-1975 53

11. Dissolved inorganic nitrogen concentrations at Station 202 in White River, 1972-1975 54

12. Dissolved inorganic nitrogen concentrations at Station 227 in White River, 1973-1975 55

13. Phosphorus concentrations at Station 201 in White River, 1972-1975 56

14. Phosphorus concentrations at Station 202 in White River, 1972-1975 57

15. Phosphorus concentrations at Station 227 in White River, 1973-1975 58

16. Silicon concentrations at Station 201 in White River, 1972-1975 . . . 59

17. Silicon concentrations at Station 202 in White River, 1972-1975 . . . 60

18. Chloride concentrations at Station 201 in White River, 1972-1975. . . 61

19. Chloride concentrations at Station 202 in White River, 1972-1975. . . 62

Figure Page

20. Location and diversion status of point sources and municipalities in the White Lake basin. 65

21. Silicon load of the White River, 1973-1975. 69

22. Cloride load of the White River, 1973-1975. 70

23. Dissolved inorganic nitrogen load of the White River, 1973-1975 71

24. Total phosphorus load of the White River, 1973-1975 72

25. Vollenweider Model relating phosphorus loading per unit of surface to hydraulic load (mean depth divided by lake retention time) for Muskegon Lakes (Vollenweider, 1975) 79

26. Dissolved inorganic nitrogen concentrations at Stations 203 and 204 in White Lake, 1972-1975. 83

27. Total dissolved phosphorus concentrations at Stations 203 and 204 in White Lake, 1972-1975. 84

28. Nitrate concentrations in White Lake, 1972-1975 86

29. Ammonia concentrations in White Lake, 1972-1975 87

30. Dissolved oxygen in the bottom waters of White Lake, 1972-1975 89

31. Soluble reactive phosphorus concentrations in White Lake, 1972-1975 91

32. Total dissolved phosphorus concentrations in White Lake, 1972-1975 92

33. Total phosphorus concentrations in White Lake, 1972-1975. 93

34. Dissolved iron concentrations in White Lake, 1972-1975. 95

35. Dissolved silicon concentrations in White Lake, 1972-1975 96

36. Chlorophyll *a* concentrations in White Lake, 1972-1975 98

37. Primary productivity rates in White Lake, 1972-1975 99

38. Correlation between chlorophyll *a* and Secchi disc (after Dillon and Rigler, 1975). 104

39. Correlation between phosphorus and summer chlorophyll *a* (after Dillon and Rigler, 1974) 105

40. Pre- and post-diversion status of selected two-meter, yearly average water quality parameters in White Lake. 106

Figure Page

41. Chlorophyll *a* concentrations - 2 November 1973 bioassay. 108

42. Chlorophyll *a* concentrations - 27 April 1974 bioassay. 109

43. Distribution of macrophytes in White Lake. 112

44. Macrophyte and phytoplankton concent in White Lake 113

45. Concentrations of nutrients in White Lake sediment interstitial waters. 115

46. Nutrient release from White Lake sediments-laboratory studies. . . 116

47. Nutrient release from White Lake sediments—*in situ* studies. . . . 118

48. White Lake phosphorus model. 120

49. Comparison of model calculations and observed data (1974) for dissolved and particulate phosphorus in White Lake 121

50. Comparison of rehabilitation schemes for White Lake. 122

51. Muskegon River average annual flow at Newaygo, Michigan (USGS Station # 04122000). 127

52. Muskegon River mean monthly flow at Newaygo, Michigan (USGS Station # 04122000). 127

53. Dissolved inorganic nitrogen concentrations in the Muskegon River, 1973-1975. 130

54. Dissolved inorganic nitrogen concentrations in the Muskegon River, 1972-1975. 131

55. Phosphorus concentrations in the Muskegon River, 1973-1975 132

56. Phosphorus concentrations in the Muskegon River, 1972-1975 133

57. Silicon concentrations in the Muskegon River, 1972-1975. 134

58. Chloride concentrations in the Muskegon River, 1972-1975 135

59. Location and diversion status of point sources and municipalities in the Muskegon Lake basin. 138

60. Silicon load of the Muskegon River, 1973-1975. 142

61. Chloride load of the Muskegon River, 1973-1975 143

62. Dissolved inorganic nitrogen load of the Muskegon River, 1973-1975. 144

Figure Page

63. Total phosphorus load of the Muskegon River, 1973-1975. 145

64. Total dissolved phosphorus concentrations at Stations 103 and 106 in Muskegon Lake, 1972-1975 154

65. Dissolved inorganic nitrogen concentrations at Stations 103 and 106 in Muskegon Lake, 1972-1975 155

66. Nitrate concentrations in Muskegon Lake, 1972-1975. 157

67. Ammonia concentrations in Muskegon Lake, 1972-1975. 158

68. Dissolved oxygen in the bottom waters of Muskegon Lake, 1972-1975 160

69. Soluble reactive phosphorus concentrations in Muskegon Lake, 1972-1975 161

70. Total dissolved phosphorus concentrations in Muskegon Lake, 1972-1975 162

71. Total phosphorus concentrations in Muskegon Lake, 1972-1975 . . . 164

72. Dissolved iron concentrations in Muskegon Lake, 1972-1975 165

73. Dissolved silicon concentrations in Muskegon Lake, 1972-1975. . . 166

74. Chlorophyll *a* concentrations in Muskegon Lake, 1972-1975. 167

75. Primary productivity rates in Muskegon Lake, 1972-1975. 168

76. Correlation between chlorophyll *a* and Secchi disc (after Dillon and Rigler, 1975). 172

77. Correlation between phosphorus and summer chlorophyll *a* (after Dillon and Rigler, 1974) 173

78. Pre- and post-diversion status of selected two-meter, yearly average water quality parameters in Muskegon Lake 174

79. Dissolved inorganic nitrogen concentrations in Black Creek, 1973-1975. 179

80. Phosphorus concentrations in Black Creek, 1973-1975 180

81. Phosphorus concentrations in Black Creek, 1972-1975 181

82. Dissolved inorganic nitrogen concentrations in Black Creek, 1972-1975. 182

83. Dissolved silicon concentrations in Black Creek, 1972-1975. . . . 183

Figure Page

84. Chloride concentrations in Black Creek, 1972-1975. 184

85. Dissolved inorganic nitrogen concentrations in Little Black Creek, 1973-1975 186

86. Total phosphorus concentrations in Little Black Creek, 1973-1975. 187

87. Location and diversion status of point sources and municipalities in the Mona Lake basin. 189

88. Dissolved inorganic nitrogen load of Black Creek, 1973-1975. 191

89. Total phosphorus load of Black Creek, 1973-1975. 192

90. Dissolved inorganic nitrogen load of Little Black Creek, 1973-1975. 193

91. Total phosphorus load of Little Black Creek, 1973-1975 194

92. Dissolved inorganic nitrogen concentrations at Stations 3 and 4 in Mona Lake, 1972-1975. 203

93. Total dissolved phosphorus concentrations at Stations 3 and 4 in Mona Lake, 1972-1975. 204

94. Nitrate concentrations in Mona Lake, 1972-1975 206

95. Ammonia concentrations in Mona Lake, 1972-1975 207

96. Dissolved oxygen in the bottom waters of Mona Lake, 1972-1975. . . . 208

97. Soluble reactive phosphorus concentrations in Mona Lake, 1972-1975. 210

98. Total dissolved phosphorus concentrations in Mona Lake, 1972-1975. 211

99. Total phosphorus concentrations in Mona Lake, 1972-1975. 212

100 Dissolved iron concentrations in Mona Lake, 1973-1975. 213

101. Dissolved silicon concentrations in Mona Lake, 1972-1975 214

102. Chlorophyll *a* concentrations in Mona Lake, 1972-1975 216

103. Primary productivity rates in Mona Lake, 1972-1975 217

104. Pre- and post-diversion status of selected two-meter, yearly average water quality parameters in Mona Lake. 221

SECTION I

SUMMARY AND CONCLUSIONS

A regional program of wastewater management went into full operation in Muskegon County, Michigan in 1974. The program consists of wastewater collection, treatment, and spray-irrigation on farmland. This system has replaced outdated inefficient treatment plants which had discharged poorly treated wastes directly to Muskegon County surface waters. The major overall goals of the project were to stimulate economic growth in the region by providing efficient waste treatment facilities at a reasonable cost, to restore water quality in three lakes important for recreation, and to meet State of Michigan water quality standards for wastewater discharges.

The research described in this report was designed to evaluate the impact of the wastewater management system on the quality of the surface waters of Muskegon County. Within this study particular emphasis was placed on an analysis of the effects of the diversion project on the trophic status and nutrient budgets of White, Muskegon, and Mona Lakes. No studies were conducted to define levels of organics, trace metals, suspended solids, pesticides, or other pollutional contaminants which may have been altered as a result of the diversion project. In this report the term water quality shall have limited scope relating primarily to parameters concerned with eutrophication and nutrient budgets.

The program of approximately biweekly lake and stream monitoring was implemented in the late spring of 1972 (the only year of the study period prior to all diversions). Diversion began in 1973 and was essentially fully implemented by 1974. White, Muskegon, and Mona Lakes were monitored at several stations and depths for phytoplankton growth nutrients, chlorophyll *a*, primary productivity, and various other physical and chemical parameters. Major tributaries to each lake were also monitored at multiple locations. These locations included the White River and Mill Pond Creek in the White Lake Basin; the Muskegon River and Bear Lake Creek in the Muskegon Lake Basin; and Little Black and Black Creeks in the Mona Lake Basin. Stations were selected to evaluate the effects of the diversion and spray-irrigation project. The results from these studies provided baseline data on limnological and stream conditions. By combining the tributary data with information on municipal and industrial nutrient loads it was possible to evaluate the potential and actual effectiveness of the diversion spray-irrigation project to reduce nutrient loads to each of the three lakes. The detailed lake monitoring program provided a suitable data base to assess changes in lake water quality in actual response of the diversion project. A lake-by-lake summary of the analysis follows.

WHITE LAKE

Baseline limnological investigations of White Lake show that prior to the diversion, the lake was highly productive and could be considered eutrophic. The lake was plagued by excessive growths of algae and nearly 20% of the lake bottom was heavily covered with macrophytes. Maximum summer chlorophyll a measurements were approximately 50 μg/ℓ and maximum primary productivity values ranged between 100 and 200 μg C/ℓ/hr. Macrophyte densities were as high as 150 grams dry wt/m^2. Secchi disc visibility was low, averaging less then 2 meters. Average surface total phosphorus concentrations were approximately 40 μgP/ℓ and bottom water concentrations often reached 100 to 200 μg P/ℓ during the summer. Winter dissolved inorganic nitrogen concentrations were observed to exceed 1 mg N/ℓ. Bottom waters were depleted of oxygen during the summer for two to three months.

The White River was found to be the major nutrient source to White Lake before diversion, with significant nutrient contributions also originating from the Whitehall Wastewater Treatment Plant and the Whitehall Leather Company. These discharges were diverted in 1973 and 1974 respectively. The calculated average pre-diversion point source nutrient loads contributed 23% of the annual total phosphorus and 31% of the annual dissolved inorganic nitrogen load to White Lake. This would be the maximum reduction in nutrient loads expected to result from the diversion project. Such a reduction would not be expected to result in a dramatic change in the trophic status of the lake.

No significant reductions in phosphorus or nitrogen river concentrations or loads were actually observed for White Lake despite the diversion. The potential reduction was believed to be obscured by relatively large fluctuations in unknown upstream loads. On the other hand, water quality in the White River was not degraded by the spray-irrigation drainage.

Detailed limnological studies were designed to examine the dynamics of spatial and temporal variation in certain water quality characteristics. Seasonal nutrient dynamics were closely tied to algal uptake, thermal stratification, hypolimnetic anoxia, bacterial decomposition, ammonification, nitrification, and allochthonous loads. Seasonal nutrient depletion due to algal and macrophytic uptake were observed for phosphorus and nitrogen. Summer hypolimnetic concentrations of nutrients were extremely high as a result of active decomposition and sediment release. Bioassay and nutrient ratio analyses characterize the spring and summer surface waters as nitrogen limited. This is confirmed by the dominance of nitrogen-fixing blue-green algae. These organisms proliferate in nitrogen poor environments while other algae which require dissolved inorganic nitrogen (ammonia, nitrite or nitrate) fail to grow. Phosphorus was determined to be the potential limiting nutrient in the fall. Detailed limnological and nutrient cycle modeling is required for the proper and detailed assessment of algal growth limitation.

Studies were conducted in White Lake to evaluate the significance of macrophytes to the total lake nutrient budget and to determine the importance of nutrient exchange between sediments and the water column to lake nutrient dynamics. These factors were considered to be important to lake chemistry and ecology and to have the potential to negate some of the beneficial effects of the nutrient diversion program.

An evaluation was made of limnological data collected over four years (1972-1975) on White Lake to determine if any improvements in water quality would be observed to result from the diversion project. Average surface concentrations of dissolved inorganic nitrogen and phosphorus were essentially unchanged between 1972 and 1975, although ammonia levels dropped slightly in the surface waters and markedly in the bottom waters while nitrate in surface waters increased. Average chlorophyll *a* levels were reduced from 12.4 µg/ℓ in 1972 to 8.6 in 1974 and 1975. These improvements were not reflected in increased water clarity. Analysis of summer hypolimnetic dissolved oxygen suggests slightly improved conditions. Bottom waters still experienced anoxia, but the rate of depletion was slower and the vertical extent somewhat less. This could be a result of decreased BOD loadings and possibly a reduction in autochthonous loads. Less severe hypolimnetic anoxia would result in reduced nutrient release from the sediments and improved conditions for benthic fauna.

Analysis of White Lake using simple models (Vollenweider, 1975; Dillon and Rigler, 1974, 1975) and a complex productivity model suggest that under 1975 loading conditions White Lake will remain eutrophic. The diversion has, however, had the effect of shifting the condition of the lake to a state whereby further reductions in nutrient loads can result in significant improvements in chlorophyll *a* levels and water transparency. Various additional management alternatives are briefly evaluated using the productivity model.

MUSKEGON LAKE

The baseline limnological conditions in Muskegon Lake were similar to White Lake. The lake was characterized by high nutrient and algal concentrations and could also be classified eutrophic. Macrophytes were present but not severely troublesome. Average chlorophyll *a* levels in 1972 were approximately 25 µg/ℓ; however, the peak level was over 50 µg/ℓ. Peak summer primary productivity was high averaging about 60 to 75 µg/ℓ. Secchi disc visibility averaged about 1.5 meters. Surface water concentrations of total phosphorus averaged 67µg P/ℓ; in 1972 however; values were measured in excess of 100µg P/ℓ. Winter dissolved inorganic nitrogen concentrations approached 400 µg N/ℓ. The bottom waters of Muskegon Lake were, like White Lake, characterized by high phosphorus and ammonia concentrations. Summer bottom water anoxia was observed; however, this condition was less stable than in White Lake. Nitrate was observed at times in summer bottom waters; suggesting the occurrence of nitrification and thus the presence of oxygen. Unstable anoxic conditions are believed to be related to storm, circulation, and ship traffic induced disruptions of the thermocline and Lake Michigan water intrusion.

The Muskegon River was found to be the major nutrient source to Muskegon Lake. Two municipal treatment plants, the S.D. Warren Paper Company, and the Storey (Ott) Chemical Company contributed significant nutrients to the lake prior to diversion. The pre-diversion point sources of nutrients accounted for 38% of the annual total phosphorus load and 50% of the annual dissolved inorganic nitrogen load. Storm water loads were estimated to contribute less than 10% of the total phosphorus load. The remaining load originates from Muskegon River upstream non-point loads.

A reduction of approximately 20 to 25% in nutrient loads (total phosphorus and inorganic nitrogen) to Muskegon Lake was actually observed subsequent to the diversion. The observed reduction was somewhat less than the anticipated reduction because of variable upstream loads. The spray-irrigation drainage did not have any consistent undesirable effects on Muskegon River water quality. The spray site load to Musekgon Lake was at most 1.5% of the total phosphorus load and 11% of the dissolved inorganic nitrogen load. Specific observations during periods of lagoon discharge and spray site fertilization are not available.

The seasonal dynamics of plankton and nutrient cycles in Muskegon Lake were similar to those observed in White Lake. Spring and summer surface nutrient depletion was caused by phytoplankton blooms. Summer hypolimnetic anoxia and bottom water nutrient accumulation were also observed. All processes were closely tied to the algal uptake, death, and decomposition cycles and thermal stratification. In Muskegon Lake nutrient ratio analyses suggest phosphorus limitation in the spring and fall. Summer waters are nitrogen limited, and as a consequence nitrogen fixing blue-green algae proliferate.

Significant changes in nutrient conditions in Muskegon Lake were observed between 1972 and 1975 following nutrient diversion. Phosphorus levels were decreased by approximately 50%. Although dissolved inorganic nitrogen was not observed to decrease, the nitrate to ammonia ratio increased favoring the more oxidized form. Average chlorophyll *a* concentrations were reduced by 62% to 9.5 μg/ℓ in 1975. These reductions did not result in increasing water clarity. Hypolimnetic dissolved oxygen levels were slightly improved. Analysis of Muskegon Lake data using the Vollenweider model show that at present loading conditions the lake will remain eutrophic. The system condition has however been shifted to a state where additional reduction in nutrients may result in observable changes in water clarity.

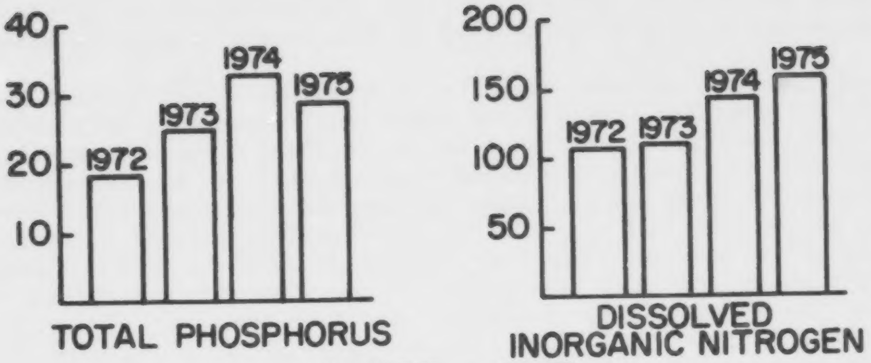
MONA LAKE

Mona Lake was the most highly productive of the three lakes. Nutrient concentrations were extremely high. Total phosphorus concentrations reached maximum values of 535 μg P/ℓ in surface waters and 2,400 μg P/ℓ in hypolimnetic waters in the summer. Dissolved inorganic nitrogen levels of 2 mg/ℓ were measured in the bottom waters. Chlorophyll *a* measurements often exceeded 100 μg/ℓ with associated primary productivities exceeding 100 μg C/ℓ/hr. Secchi disc visibility averaged 1 meter in depth with a minimum value of 0.3 meters observed. The bottom waters of the lake were devoid of oxygen for as long as four months during the summer.

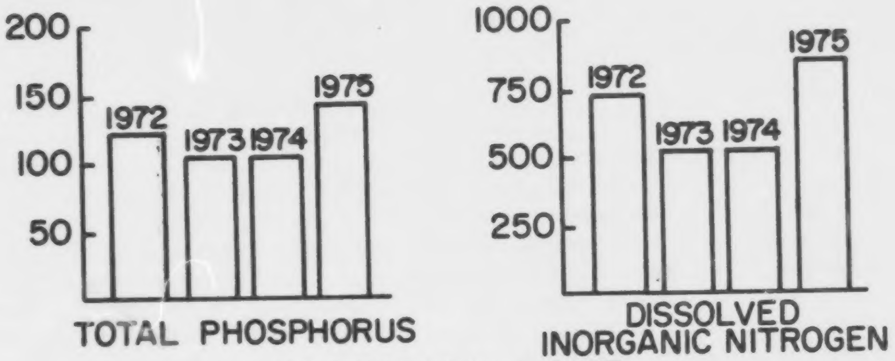
Black Creek was the most significant hydrologic input to Mona Lake, however, prior to diversion Little Black Creek contributed more nutrients. The principal sources of nutrients to Little Black Creek were municipal and industrial discharges as well as urban runoff. It was estimated that the prediversion point source loads represented 65% of the total phosphorus load to the lake and 45 to 65% of the dissolved inorganic nitrogen load. Since all of the point sources were not diverted by 1975 the potential reduction in load as a consequence of the diversion project was 60% for total phosphorus and 40 to 55% for dissolved inorganic nitrogen. The actual 1972 to 1975 observed reduction was 65% for phosphorus and 30 to 45% for dissolved inorganic nitrogen. Nutrients loads from storm runoff, urban drainage, and other unidentified sources were considered significant. Spray site drainage did not have a dramatic effect on Black Creek water quality. The nutrient loads from the drainage were 5 to 15% of the total post diversion load.

Algal nutrient uptake, decomposition and sediment release, thermal stratification, nitrification, and denitrification were observed to affect the seasonal dynamics of limnological cycles in Mona Lake. The processes were quite similar to those observed in White and Muskegon Lakes. However, Mona Lake algal dynamics were effected by the application of algicides in the summer. As a consequence, the maximum phytoplankton population occurred in the late fall in contrast with the summer maximum observed in the other two lakes. Nutrient ratio analyses indicate that Mona Lake has a potential for nitrogen limitation. Nitrogen fixing blue-green algae were not observed to dominate in Mona Lake. All nutrient concentrations were excessively high in Mona Lake waters and actual nutrient limitation of growth was not expected.

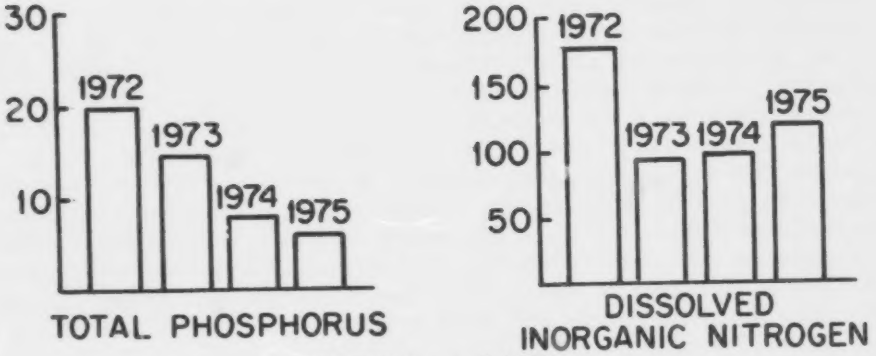
Of the three lakes studied, Mona Lake experienced the largest reduction in average surface water phosphorus concentrations (60-70%). Summer bottom water ammonia and phosphorus concentrations also were reduced dramatically. Average surface water nitrogen concentrations increased slightly. However, chlorophyll *a* concentrations were measured to be markedly higher all three years following the diversion. This trend could be the result of inconsistent year-to-year applications of algicides. No improvements in water clarity were observed. Application of simple eutrophication correlations suggest that further reductions in nutrients are required before substantial improvements in chlorophyll *a* and Secchi disc can be realized. Descriptions of changes in loading rates and nitrogen, phosphorus and chlorophyll *a* concentrations are presented in Figures 1 and 2.



WHITE LAKE



MUSKEGON LAKE



MONA LAKE

Figure 1. Annual average nutrient loadings to the Muskegon Lakes.

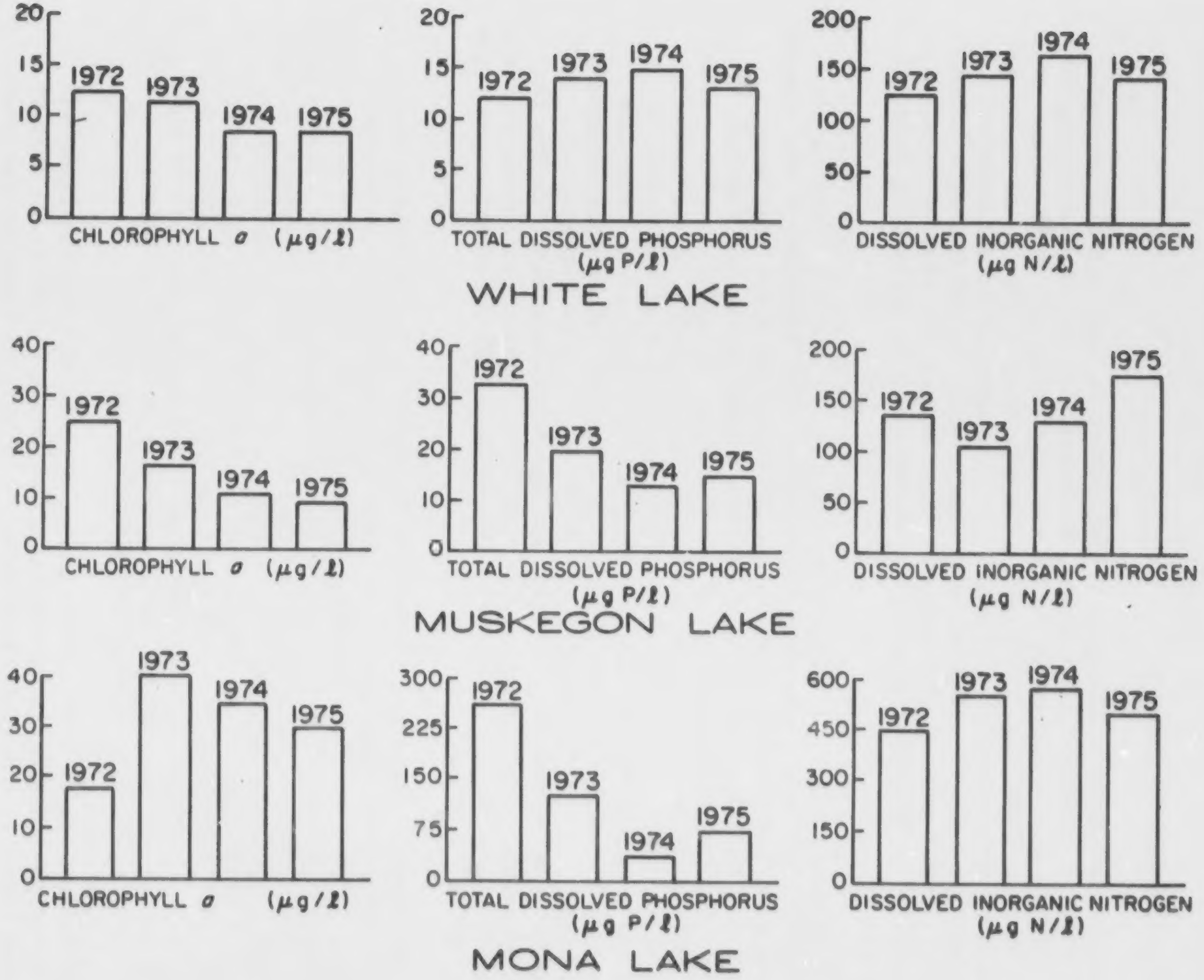


Figure 2. Annual average surface water concentrations of selected water quality variables in the Muskegon Lakes.

SECTION II

RECOMMENDATIONS

FUTURE WATER QUALITY STUDIES

This study represents a thorough investigation of nutrient and eutrophication related parameters for White, Muskegon, and Mona Lakes during the years 1972 to 1975. Extensive data were also collected on tributary nutrient loads. There were, however, inherent limitations to the study and as a consequence, additional research is necessary to more completely understand the factors which affect water quality in the Muskegon Lakes. The major limitations of this report evolve around its limited duration and scope. The following studies are recommended in descending order of priority.

First, it is important to continue to monitor the water quality in the three lakes and their associated tributaries. The duration of the present study was not sufficiently long to accurately observe the complete response of the lakes following diversion. This limitation resulted from the delayed implementation of the diversion spray-irrigation system which shortened the actual post-diversion study period. An additional two or three years of monitoring is recommended so that the extent of additional improvements in water quality can be determined. Adequate information is presently available on the seasonal and spatial dynamics of the lakes. A limited sampling program is therefore recommended. Two stations should be sampled for each lake on a biweekly basis during the ice-free season. Monitoring should focus on nutrients (including kjeldahl nitrogen), chlorophyll a, dissolved oxygen, biochemical oxygen demand, and primary productivity.

A similar biweekly sampling program is recommended for tributary monitoring. Additional special studies should be conducted to accurately quantify the sources of non-point contamination to each of the tributaries and determine which of these sources are controllable. These studies must include an assessment of storm sewer runoff, waste lagoon seepage, illegal discharges, urban runoff, and septic field drainage. Estimates of the importance of some of these sources were presented within this report although detailed assessments were beyond the scope of the study. Data from this study indicate that additional dramatic improvements in the trophic status of the Muskegon Lakes can only be achieved through control of these non-point and unidentified nutrient sources.

A major limitation of this report is the absence of complete taxonomic analysis of phytoplankton, zooplankton, and benthos samples to support the other chemical and biological data. However, preliminary analyses of these data generally support the conclusions of this report (Meier, 1977). It is strongly recommended that these samples be thoroughly analyzed. Continued sampling of phytoplankton, zooplankton, and benthos is also recommended. Changes in species composition detected through taxonomic analysis can often identify alterations in water quality which may not otherwise be observed.

It is recommended that additional mathematical water quality models be developed for Muskegon and Mona Lakes. These should be of the same level of sophistication as the White Lake model. Development of these models would permit more accurate, quantitative, and reliable evaluation of basin and water management alternatives and their effects on lake water quality. Use of such models will permit necessary cost-effectiveness evaluations.

A fourth recommendation for further research is to monitor various trace organic and trace metal contaminants in the lakes. Although adequate data are not available from a pre-diversion period, information may be obtained by examining sediment concentrations and correlating sediment depth with time. These investigations would permit an evaluation of trace contaminant water quality problems in the lakes and could relate possible changes to the diversion spray-irrigation system or other basin management practices.

The last recommendation is for detailed process studies to evaluate macrophyte ecology, sediment water interactions, epilimnetic and hypolimnetic mixing, and algal growth potential in each of the lakes. These processes have been found to be of varying importance in the lakes and need to be further studied for accurate scientific understanding and rational management.

MANAGEMENT RECOMMENDATIONS

Studies on White Lake have demonstrated that the major source of nutrient contamination originates from unidentified sources in the White River upstream of the land disposal site. If further significant water quality (nutrient and chlorophyll *a*) improvements are desired, then these non-point sources must be identified, quantified and controlled where feasible. This may require land-use management. Particular emphasis should be focused on nutrient contamination from septic tank seepage, celery farm drainage, and other sources of agricultural and forest run-off.

In Muskegon Lake, upstream nutrient sources were also determined to be dominant. The control of the sources may require land use management. First, however, more detailed quantification of the sources is required. A limited (up to 10%) reduction in nutrient loads to Muskegon Lake may be achieved by controlling storm-sewer runoff. Contamination of the lake by substances not studied in this research may have changed as a result of the diversion, but additional urban sources may have to be controlled to achieve further water quality improvements. Future studies will help answer these management questions.

In the Mona Lake basin the most significant and immediate reduction in remaining nutrient loadings would result from elimination of urban and storm runoff, and industrial discharges to the Little Black Creek Basin. This is the highest priority. Waste lagoon seepage to Black Creek may also be an important nutrient source. Only after control of these sources should the difficult task of land managment be addressed.

Direct lake rehabilitation programs such as dredging, aeration, and mixing are not recommended for any of the lakes unless further reductions in nutrient loads are achieved. Macrophyte harvesting and limited sediment dredging may be helpful in White Lake to improve accessibility. However, macrophytes represent a habitat for many lake fish and should not be entirely eliminated. Full lake bottom dredging in White Lake is not recommended. Detailed data concerning the effects of sediments and macrophytes were not obtained for Muskegon and Mona Lakes. However, lake bottom sediment dredging of Muskegon and Mona Lakes is not recommended unless significant future nutrient load reductions are achieved. Macrophytes are not a problem of sufficient magnitude in either Muskegon or Mona Lakes to require harvesting.

SECTION III

INTRODUCTION

Muskegon County, Michigan has recently been the focus of a considerable amount of national attention. The region has had a history of uncontrolled exploitation of its natural resources. Decades of resource degradation, industrial sprawl, uncontrolled development, and pollution have forced the county into a recent economic crisis. The county has now taken an aggressive and innovative step in wastewater management. The county implemented a wastewater management program whereby numerous municipal and industrial waste discharges that were formerly discharged directly or indirectly to Muskegon, White, and Mona Lakes are now diverted and transported to an inland spray-irrigation and disposal site. The wastes are treated with secondary aeration in lagoons. The wastewater is then sprayed on land where agricultural crops and soil act to filter and cleanse the effluent through what has been termed a "living filter." This program is based on the environmental concept that air, water, and land are one interacting system. No wastes can be discarded without ultimate repercussions and these wastes represent misplaced resources.

The concept of spray-irrigation of wastes to dispose of sewage and simultaneously raise farm crops is not new to the United States. Muskegon County, however, was the first in the United States to attempt such a system on a large scale to a regional population. The chief goal of the system is to eliminate pollutional discharges efficiently and economically. It is also hoped that implementation of the system would restore the natural beauty of the lake environments degraded by pollution thus promoting industrial growth and recreational development in the area.

A multidisciplinary research effort has been directed towards evaluating the effectiveness of the new waste management system. This effort has involved scientists and investigators from the U.S. Environmental Protection Agency, the University of Michigan, Michigan State University, the Michigan Water Resources Commission, and Muskegon County. As part of this program, the University of Michigan has been studying the impact of the spray-irrigation system on surface water quality in the region. White, Muskegon, and Mona Lakes are the most largely exploited lakes of the region, and thus were the focus of the study. The objectives of this study were:

1. To document pre-diversion water quality in the three Muskegon County lakes and their tributaries.
2. To evaluate potential and actual reductions in nutrient loads to the lakes following implementation of the sewage diversion, spray-irrigation system.

3. To assess potential and actual improvements in lake water quality resulting from the implementation of the system. In this study the water quality focus was on phytoplankton growth nutrients, primary productivity, and eutrophication related parameters.

SECTION IV

DESCRIPTION OF STUDY AREA AND SOCIO-ECONOMIC HISTORY

Muskegon County is located on the eastern shore of Lake Michigan near the midpoint of the western side of the lower peninsula in the State of Michigan (see Figure 3). Muskegon County has a population of 157,000 which is principally concentrated in two urban areas. The larger of the two areas is the Muskegon, Norton Shores, North Muskegon, and Roosevelt Park region which includes 80% of the population. These municipalities are located in an area surrounding Mona and Muskegon Lakes. Muskegon and Muskegon Heights are industrialized. The prominent industries are the S.D. Warren Company (paper mill); the Storey (Ott) Chemical Company; and various other chemical, manufacturing, and foundry-related industries. The other significant population center is Whitehall and Montague located at the eastern end of White Lake. This area is much less industrialized. The only significant industries located there are the Whitehall Leather Company, the Hooker Chemical Company, and the Misco Division of Howmett Corporation (heavy manufacturing). The lands outside of these urban areas, which include the remainder of the drainage basins of the three lakes, are primarily forested and rural.

The surface geology of the region is principally a reflection of glacial effects. The surface features consist of glacial lake beds, outwash, end moraines, ground moraines, sand dunes, delta deposits, abandoned beaches, and waterlaid moraines. These features can be categorized into four types: 1) moraine uplands, 2) outwash plains, 3) glacial lake plains, and 4) alluvial lowlands. The moraine uplands are located in the eastern and northwestern part of the county and are generally associated with terminal moraines. The outwash plains are located west of the uplands and exhibit an irregular and undulating topography. The glacial lake plain is in the west and central portion of the county, while the alluvial lowlands are associated with natural stream cuts in the terrain. The glacial lake beds were formed primarily during the Glenwood stage of Lake Chicago. The Muskegon and White Rivers are modern counterparts of preglacial streams that flowed into Lake Chicago and deposited sand and gravel to form deltas. Because of the sand and gravel content of the surface glacial deposits, water infiltration rates of the soil are high. The groundwater table in the area is also very high, often only 5 or 10 feet from the surface. During extremely wet periods or when lake levels are high, groundwater can cause basement flooding, septic tank malfunction, and create marsh and wetland areas.

The climate of the region is typically midwestern although somewhat modified by Lake Michigan. The area receives an average annual precipitation of 79.8 centimeters. The dominant winds are westerly. During

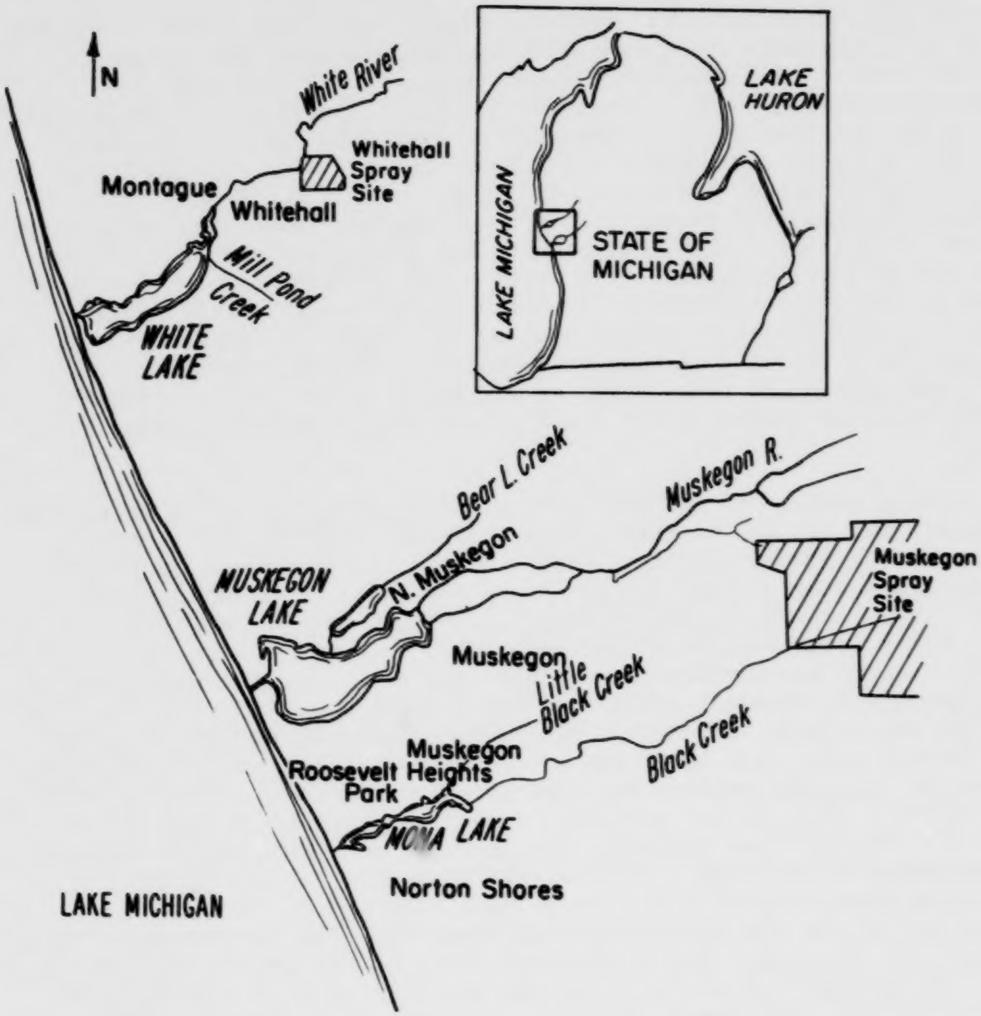

N
White River
Whitehall Spray Site
Montague
Whitehall
Mill Pond Creek
WHITE LAKE
LAKE HURON
LAKE MICHIGAN
STATE OF MICHIGAN
Bear L. Creek
Muskegon R.
N. Muskegon
MUSKEGON LAKE
Muskegon Spray Site
Muskegon
Little Black Creek
Black Creek
Muskegon Heights
Roosevelt Park
MONA LAKE
Norton Shores
LAKE MICHIGAN

Figure 3. The Muskegon Lakes study area.

winter months warm westerly winds bring significant amounts of precipitation. This is reflected by relatively high winter flows observed in rivers, a consequence of snow melt and rain.

Muskegon County became a prosperous center of urban activity in 1840. During the nineteenth century the economy was based almost entirely on the exploitation of the valuable pine forests of the area. Rivers cut deep into the forest providing cheap transportation of logs and the dune impounded lakes at the river mouths provided excellent sites for mills. Unfortunately, a half century of clear-cutting practices stripped the area of its forest resources. At approximately the same time industrialization was blooming in the region. Easy access to Great Lakes water routes and railroad transportation, as well as an abundant supply of foundry sands, encouraged heavy industry. Markets for the goods produced were first the expanding railroads, followed by the auto industry, and then national defense contracts—a vast market heavily maintained by two world wars. Another natural resource that boosted the economy of the area was oil, first discovered in 1927. By 1940, however, the wells were no longer profitable. Most oil reserves had been depleted due to ravenous exploitation, inadequate regulation, and inefficient extraction. Air and water pollution grew worse in the area as urban and industrial development grew unchecked and discharged wastes to the environment.

Attempts to introduce farming in 1930 were not highly successful. The land had been left barren and was unsuited for farming because the soils were very sandy and had little topsoil. As a consequence, recession struck heavily in the area in the 1950's and 1960's. By 1968 the unemployment rate was double the national average and per capita income was the third lowest in the state. Population growth in the county between 1960 and 1970 was 4.2% compared to 13.4% for the entire state. Industries could no longer compete with more modern plants elsewhere and environmental degradation limited expansion. New industry was not attracted to the area because of the largely unskilled work force and pollution problems. Tourism, the third largest industry of the county, was also threatened by air and water pollution, exploitation of the coastal sand dunes, and uncontrolled urban sprawl.

As a result, Muskegon County took innovative steps to divert all waste discharges from the lakes and pipe them inland for treatment and spray-irrigation on the land. It was hoped that this scheme would meet the increasingly strict government regulations on pollutional discharges and would revitalize the polluted Muskegon County environment. The plan was based on three considerations:

1. The environment is one system with interactions among air, land, and water. Cultural development affects and is affected by the system.

2. The system is considered closed for planning purposes. Nothing can be thrown away that could return to trouble us later.

3. Wastewaters are potential resources out of place. If relocated in the environment, they can take on new values.

The goals and objectives of the diversion spray-irrigation project were:

1. To protect Lake Michigan from materials that stimulate excessive algal growth, toxic materials, and other contamination.

2. To eliminate present sources of pollution in the prime shoreline lakes (Muskegon, Mona, and White).

3. To maximize environmental improvement and facilitate economic development, tourism, and other water-oriented recreational activities.

4. To encourage programs that provide opportunities to reclaim the large areas of unproductive land lying dormant in the county.

5. To facilitate regional economic development by providing environmental management systems with long-range economic efficiencies and required performance capabilities.

SECTION V

HISTORY AND OPERATION OF THE MUSKEGON DIVERSION SPRAY-IRRIGATION SYSTEM

The Muskegon wastewater management system consists of six basic components:

1. Collection and transportation of wastewater to the inland treatment site.
2. Biological treatment of the wastes in aerated lagoons.
3. Storage of the treated wastes in holding basins.
4. Spray-irrigation of the treated wastewater on the land.
5. Filtration of the wastewater through the soil and uptake by crops.
6. Collection of the drainage by underdrains and subsequent discharge to receiving streams.

The county has established two separate systems to handle wastes. The larger of the two sites handles the wastes formerly discharged to Muskegon and Mona Lakes. It presently accepts approximately 28 MGD and has a 1992 design flow of 42 MGD. The drainage from this spray system is discharged to both the Muskegon River via Mosquito Creek and to Black Creek. A smaller system handles the wastewater formerly discharged to White Lake. This 1.4 MGD system discharges its spray drainage to the White River.

Construction of the system began in 1971 although full operation did not begin until 1974. A detailed discussion of the history of the development and operation of the system is available elsewhere (Muskegon County, 1976). Wastes from the communities of Muskegon, Norton Shores, and Roosevelt Park were diverted on May 10, 1973. This eliminated the Muskegon Wastewater Treatment Plant discharge to the Muskegon River and a portion of the Muskegon Heights Wastewater Treatment Plant discharge to Mona Lake. On May 30, 1973 Muskegon Heights diverted the remainder of their discharge. The City of Whitehall diverted their municipal wastes from White River to the spray site on July 18, 1973. Also on June 4, 1973 the S.D. Warren Company diverted its wastewater discharge from Muskegon Lake. In 1974 Storey (Ott) Chemical Company diverted its wastewater discharge from the Muskegon River (April 18) and on November 24 the Whitehall Leather Company diverted its waste from direct discharge to White Lake. Other industries have or are in the process of diverting their waste discharge to the spray system. However, many areas surrounding the Muskegon Lakes are still unsewered and/or not connected. The status of areas surrounding each lake is given in Figures 20, 59 and 87 and Appendix F. A chronology of wastewater diversion is presented in Table 1.

In 1973 wastes were treated and stored in the large basins until 1974 when spraying of the wastes began. However, problems with clogging of spray-irrigation equipment and waste transmission pipes, and electrical and construction difficulties prevented full and regular spraying during all of 1974. As a consequence, 247 million gallons of pretreated and stored wastes were spilled directly into Mosquito Creek during the summer of 1974 and 2,263 million gallons were again discharged directly without spraying during the winter of 1974 to 1975. Since that time, the system has performed well according to design. During 1975, 7,437 million gallons of waste were sprayed on the Muskegon-Mona site. The drainage discharged 6,659 and 1,703 million gallons to the Muskegon River (Mosquito Creek) and Black Creek, respectively. The differences in the water balance are due to rainfall and natural drainage. Nitrogen fertilizers were used on the irrigated crops during 1974 and 1975. The efficiency of natural filtration and plant uptake to remove wastes and nutrients was similar to the design expectations: 99% biochemical oxygen demand removal, 99% suspended solids removal, 90% phosphorus, and 76% nitrogen removal. The monitored nutrient content of the Mosquito Creek drainage during 1975 was 50 μg P/ℓ total phosphorus, 600 μg N/ℓ ammonia, 2,000 μg N/ℓ nitrate, 3 mg/ℓ BOD_5, and 7 mg/ℓ suspended solids.

Table 1. CHRONOLOGY OF SOME IMPORTANT EVENTS RELEVANT TO THE MUSKEGON COUNTY WASTEWATER TREATMENT PROJECT

Date	Event
May 10, 1973	Muskegon municipal wastes are diverted to the spray-irrigation facility.
May 10, 1973	Norton Shores municipal wastes are diverted to the spray-irrigation facility.
May 10, 1973	Muskegon Heights municipal wastes are diverted to the spray-irrigation facility.
May, 1973	Begin storage of diverted waste at Muskegon site.
June 9, 1973	North Muskegon municipal wastes are diverted to the spray-irrigation facility.
July 18, 1973	City of Whitehall municipal wastewaters are diverted to the spray-irrigation facility.
July 18, 1973	Misco Division of Howmett Corporation (Whitehall) industrial wastewaters are diverted to the spray-irrigation facility.
June, 1974	Spray irrigation begins with erratic operation.
June, 1974	Brown and Morse industrial wastewaters are diverted to the spray-irrigation facility.
June, 1974	Johnson Products wastewaters are diverted to the spray-irrigation facility.
November 24, 1974	Whitehall Leather industrial wastewaters are diverted to the spray-irrigation facility.
November, 1974	Spray irrigation halted.
December, 1974	Direct discharge from storage lagoons begins.

Table 1.--continued

Date	Event
February, 1975	Direct discharge from storage lagoons ends.
April 15, 1975	Spray-irrigation begins again with consistent operation.
November, 1975	Spray-irrigation season ends.

SECTION VI

METHODS AND MATERIALS

GENERAL

A program was devised to evaluate background water quality in Mona, Muskegon, and White Lakes and to evaluate changes following nutrient diversion. This study consisted of a comprehensive limnological sampling program which monitored physical, chemical, and biological parameters in the lakes and their tributaries. Water quality sampling began in the spring of 1972 with an abbreviated sample frequency and station schedule. From 1973 to the fall of 1975 the sampling program was at full intensity. A summary of the sampling cruise schedule is given in Table 2. In general, the program consisted of biweekly sampling of lake and tributary waters during the ice-free season and occasional sampling during periods of ice cover. This sampling permitted an assessment of lake water quality with respect to average annual and seasonal dynamics. Although weekly or twice weekly sampling might have provided better definition of the dynamics of the system, the expense of such an effort could not be justified in view of the limited goals and funds of the program. The biweekly schedule offered a reasonable compromise between expense and sampling intensity which provided adequate information on seasonal dynamics and quantification of season tributary loads.

The location of the lake sample stations are shown in Figures 4, 5, and 6. The establishment of two sample collection stations in both Mona and White Lakes was considered adequate for the assessment of average lake water quality and for the evaluation of spatial variations. In Muskegon Lake four lake stations were sampled. The additional stations were considered appropriate because of the larger size of the lake and because of the diversity of land usage and tributary, municipal, and industrial discharges. One station was added at the channel outlet of each lake. This station provided information regarding the exchange of waters (and their associated quality) between the individual lakes and Lake Michigan.

In conjunction with the lake sampling program all major tributaries to the three lakes were sampled. The sample station locations are shown in Figure 7 and described in Table 3. The design of this sampling scheme was selected to provide information on chemical loading to the lakes and to provide information regarding the source of such loadings. Sample stations were located to determine the effect of the wastewater diversion and spray-irrigation system on the lakes.

Table 2. SAMPLING CRUISE SCHEDULE

Cruise No.	Dates	Year	Cruise No.	Dates	Year
1	May 23-24	1972	33	March 18-21	1974
2	June 21-22	1972	34	April 9-13	1974
3	July 5-6	1972	35	April 22-26	1974
4	July 19-20	1972	36	May 6-10	1974
5	August 13-16	1972	37	May 21-24	1974
6	August 28-31	1972	38	June 4-6	1974
7	September 16	1972	39	June 13-20	1974
8	September 30-October 1	1972	40	July 1-4	1974
9	October 24	1972	41	July 15-18	1974
10	November 20-22	1972	42	August 13-16	1974
11	January 16	1973	43	August 26-28	1974
12	March 5-6	1973	44	September 16-19	1974
13	March 26-28	1973	45	October 8-10	1974
14	April 12-14	1973	46	October 22-24	1974
15	April 24-27	1973	47	November 6-9	1974
16	May 8-12	1973	48	November 19-21	1974
17	May 23-25	1973	49	December 9-12	1974
18	June 11-14	1973	50	January 27-29	1975
19	June 26-29	1973	51	February 24-27	1975
20	July 10-12	1973	52	March 26-27	1975
21	July 24-26	1973	53	April 7-10	1975
22	August 8-10	1973	54	April 21-25	1975
23	August 29-September 2	1973	55	May 5-8	1975
24	September 11-14	1973	56	May 20-23	1975
25	September 26-27	1973	57	June 4-6	1975
26	October 8-11	1973	58	June 16-19	1975
27	October 23-26	1973	59	July 15-17	1975
28	November 6-8	1973	60	July 28-30	1975
29	November 19-21	1973	61	August 11-13	1975
30	December 11-13	1973	62	August 26-29	1975
31	January 21-24	1974	63	September 8-10	1975
32	February 25-28	1974	64	October 20-22	1975

Table 3. LOCATION OF TRIBUTARY SAMPLING STATIONS

Station Number	Tributary Name	Lake System	Location	Comment
2	Black Creek	Mona	Sullivan Road	Downstream of spray drainage
25	Black Creek	Mona	Swanson Road	Within the spray drainage region; added in 1973
26	Little Black Creek	Mona	Hoyt Street Bridge	River mouth; added in 1973
27	Black Creek	Mona	Bus. Rte. 31 Bridge	River mouth; added in 1973
102	Muskegon River	Muskegon	Mill Iron Road	Downstream of spray drainage
143	Muskegon River	Muskegon	Maple Island Road	Upstream of spray drainage; added in 1973
144	Muskegon River	Muskegon	Bus. Rte. 31	North branch near the river mouth; added in 1973
145	Muskegon River	Muskegon	Middle Branch	Abandoned during 1974
146	Muskegon River	Muskegon	South Branch	Abandoned during 1974
147	Muskegon River	Muskegon	At Confluence of Middle & South Branches	Sampled from boat; added in 1973
148	Bear Lake Channel	Muskegon	Ruddiman Bridge	Near creek mouth; added in 1973
201	White River	White	Fruitvale	Upstream of spray drainage
202	White River	White	US 31	Downstream of spray drainage
226	Mill Pond Creek	White	At Pond Falls	Near creek mouth; added in 1973
227	White River	White	Bus. Rte. 31	Near river mouth; sampled from boat; added in 1973

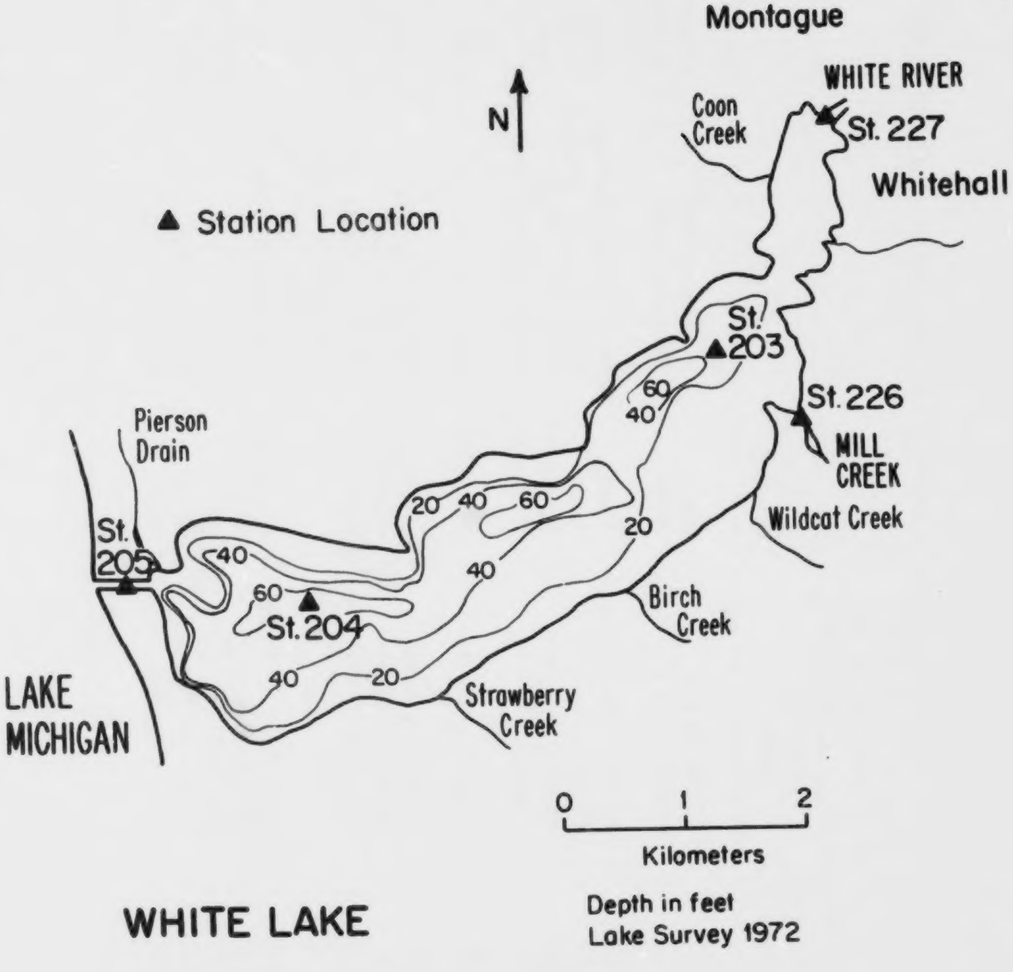

Montague
WHITE RIVER
St. 227
Coon Creek
N
Whitehall
▲ Station Location
St. 203
60
40
St. 226
MILL CREEK
Pierson Drain
Wildcat Creek
20
40
60
St. 205
40
60
St. 204
40
Birch Creek
LAKE MICHIGAN
40
20
Strawberry Creek
0 1 2
Kilometers
WHITE LAKE
Depth in feet
Lake Survey 1972

Figure 4. White Lake sampling stations.

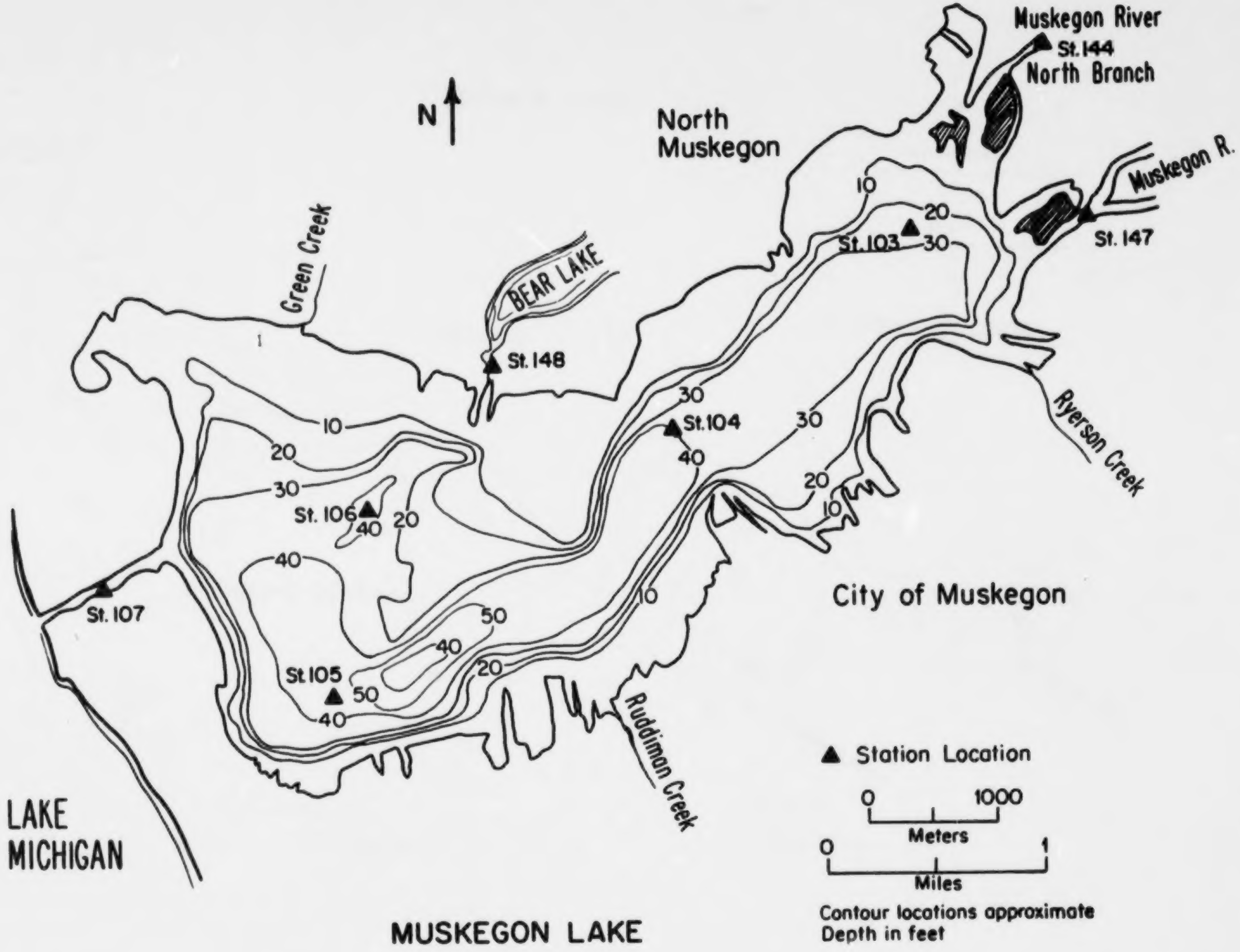

N
North Muskegon
Muskegon River
St. 144
North Branch
Muskegon R.
St. 147
Green Creek
BEAR LAKE
St. 148
St. 103
St. 104
St. 106
St. 105
St. 107
10
20
30
40
50
Ryerson Creek
Ruddiman Creek
City of Muskegon
LAKE MICHIGAN
MUSKEGON LAKE
▲ Station Location
0
1000
Meters
0
1
Miles
Contour locations approximate
Depth in feet

Figure 5. Muskegon Lake sampling stations.

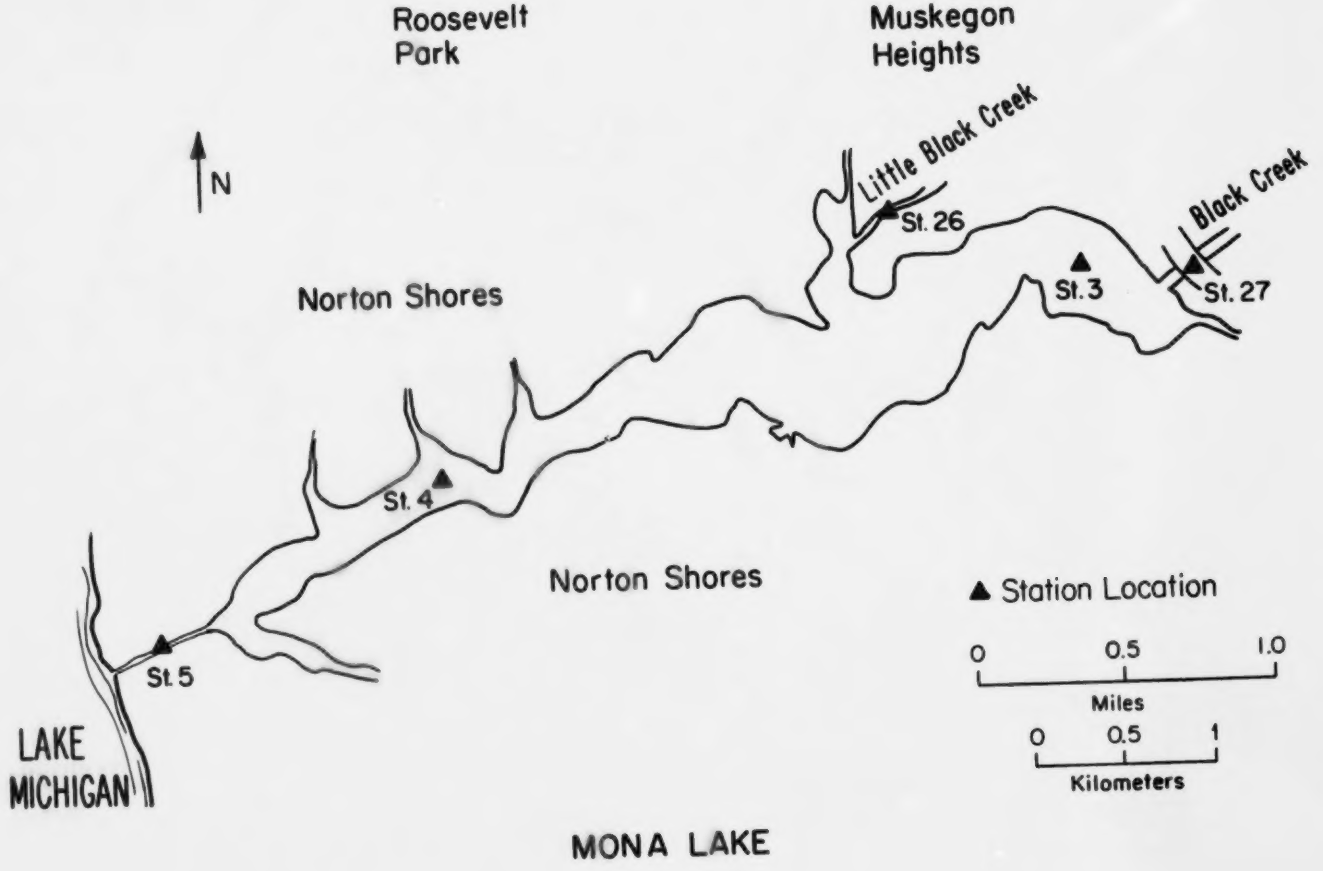

Roosevelt Park
Muskegon Heights
Little Black Creek
Black Creek
N
St. 26
St. 3
St. 27
Norton Shores
St. 4
Norton Shores
St. 5
LAKE MICHIGAN
▲ Station Location
0
0.5
1.0
Miles
0
0.5
1
Kilometers
MONA LAKE

Figure 6. Mona Lake sampling stations.

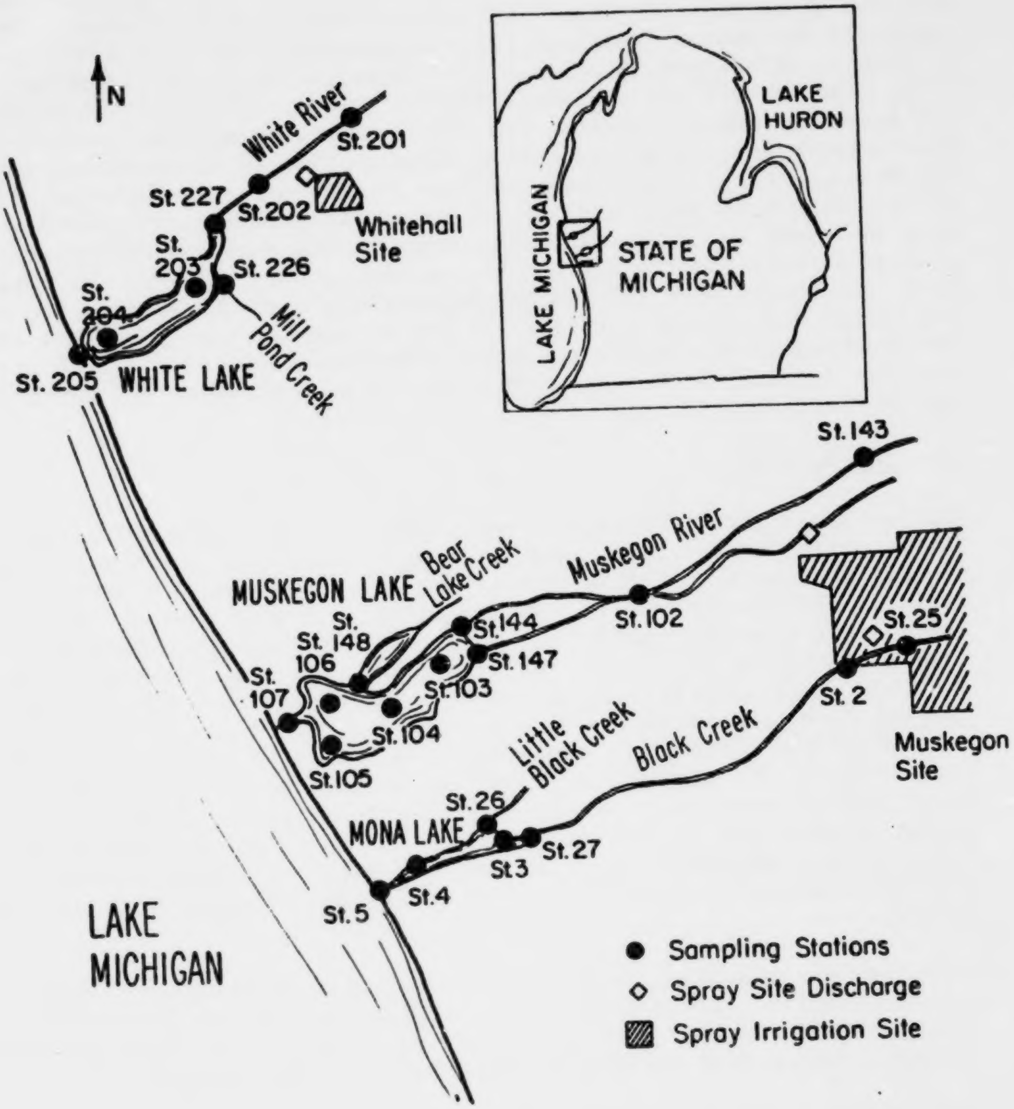

N
White River
St. 201
St. 227
St. 202
Whitehall Site
St. 203
St. 226
St. 204
Mill Pond Creek
St. 205
WHITE LAKE
LAKE HURON
LAKE MICHIGAN
STATE OF MICHIGAN
St. 143
Muskegon River
MUSKEGON LAKE
Bear Lake Creek
St. 148
St. 144
St. 102
St. 106
St. 147
St. 107
St. 103
St. 104
St. 105
St. 25
St. 2
Muskegon Site
Little Black Creek
Black Creek
St. 26
MONA LAKE
St. 27
St. 3
St. 4
St. 5
LAKE MICHIGAN
● Sampling Stations
◇ Spray Site Discharge
▨ Spray Irrigation Site

Figure 7. Tributary and lake sampling stations.

Three sample stations were located on the White River and on Mill Pond Creek. No other tributaries merited monitoring. The stations on the White River were located at the mouth (Station 227) and upstream (Station 201) and downstream (Station 202) of the spray-irrigation site. The Whitehall Wastewater Treatment Plant discharged a short distance upstream of Station 227 before diversion. Two Muskegon Lake tributaries were considered of sufficient significance to sample: The Muskegon River and Bear Lake Creek. Station 148 was located on Bear Lake Creek. The Muskegon River stations were located on both of the branches entering Muskegon Lake. Stations were also located upstream (Station 143) and downstream (Station 102) of the spray-irrigation site. Pre-diversion municipal and industrial discharges to the Muskegon River were located between Station 102 and the mouth. Sample stations for Mona Lake tributaries were located at the mouth of Black Creek (Station 27) and Little Black Creek (Station 26) and at a site just downstream of the discharge from the spray site to Black Creek (Station 2). A station was also located in close proximity to the area at which the spray drainage is discharged (Station 25).

Sampling Methods

Routine samples were obtained from aboard a 21-foot outboard Boston Whaler which was trailered between lakes. Occasionally special sampling cruises were conducted from the University of Michigan R/V MYSIS. This included cruises for sampling lake sediment. During periods of ice cover an all-terrain vehicle with a plastic dinghy in tow provided transportation on the lakes. Water samples were taken from the lake stations using a four-liter polyvinyl chloride Van Dorn sampling bottle. Samples were taken at all lake stations at 2 meters and at a depth 1 meter from the bottom. At one station in each lake (Station 4 in Mona Lake, Station 106 in Muskegon Lake, and Station 204 in White Lake) water samples were taken at a 1-meter depth and at mid-depth approximately midway between the surface and bottom. The bottom water samples were taken using a modified, inverted Van Dorn bottle designed to close when a weight suspended 1 meter below the bottle made contact with the lake bottom.

Tributary water samples were taken from surface waters only. When access permitted, river mouths were sampled from the Boston Whaler using Van Dorn sample bottles. Otherwise, samples were taken with plastic buckets lowered from bridges or cast from the river banks.

Water samples were siphoned from the Van Dorn bottles or sampling buckets into necessary containers for sample storage and preservation. A small laboratory was located in Whitehall, Michigan where samples were processed and prepared for transport to University of Michigan laboratories for analysis. Specific methods of sample storage and preservation are discussed in the next subsection on techniques of analysis.

In general, samples were collected and then transferred to 1-liter polyethylene containers with the appropriate preservatives added. Waters intended for nutrient and ion analysis were temporarily stored in ice water without preservatives. Aliquots were transferred to smaller polyethylene bottles at the Whitehall laboratory and frozen for transport to Ann Arbor. Other portions were pressure filtered through 0.45 μ acid-washed Millipore filters and then frozen. Other operations performed at the Whitehall laboratory included only those activities not compatible with sample storage or later analysis. These included filtration of chlorophyll *a* samples and titration of dissolved oxygen samples.

The methods of analysis for water samples generally conformed to techniques as described in Standard Methods (APHA, 1971). The methods of analysis used are summarized in Table 4 along with their associated uncertainty at concentrations typically measured in the Muskegon Lakes. A detailed description of the physical, chemical, and biological methods used is given in the following sections. Unless otherwise noted all chemical analyses were performed on a Technicon Model II AutoAnalyzer.

Quality control tests were routinely performed. Periodically samples having a known concentration were measured alongside lake water samples. The frequency of occurrence was approximately once for each twenty-five lake samples. Spike and recovery experiments were also routinely performed. Studies were conducted to test storage and preservation techniques. Results from all of the quality control procedures indicate satisfactory precision and accuracy. In 1973 the analytical staff participated in a U.S. Environmental Protection Agency interlaboratory nutrient analysis comparison. The results of this study are shown in Table 5. An additional analytical comparison study was conducted between the University of Michigan laboratories and the Michigan Department of Natural Resources Water Resource Commission laboratories. Collectively the above studies indicate high reliability of the University of Michigan analysis techniques.

PHYSICAL AND CHEMICAL METHODS OF ANALYSIS

Alkalinity

Unfrozen samples were stored in completely filled polyethylene bottles. Analyses were completed immediately following the cruise using acidification - pH measurement techniques (Strickland and Parsons, 1968). The samples were titrated with 0.01 N hydrochloric acid to a pH below 4.

Biochemical Oxygen Demand

Biochemical oxygen demand (BOD) was measured on a limited number of samples from White Lake. Samples were stored under refrigeration in

Table 4. SUMMARY OF METHODS OF ANALYSIS

Parameter	Method	Remarks
Alkalinity	Acid titration	Uncertainty: 0.02 meq/ℓ
Ammonia Nitrogen	AutoAnalyzer phenate reaction	L_D: 10 µg N/ℓ Uncertainty: 15 µg N/ℓ
Biochemical Oxygen Demand	Incubation Standard Methods	L_D: 0.15 mg/ℓ Uncertainty: 20%
Chloride	AutoAnalyzer thiocyanate method	L_D: 0.2 mg/ℓ Uncertainty: 3%
Chlorophyll *a*	Fluorometry	L_D: 0.01 µg/ℓ Uncertainty: 8%
Chemical Oxidation Demand	Dichromate oxidation	L_D: 1 mg/ℓ Uncertainty: 10%
Dissolved Oxygen	Azide-Winkler titration or potentiometric electrode	Uncertainty: 0.05 mg/ℓ
Iron	AutoAnalyzer TPTZ method	L_D: 20 µg/ℓ Uncertainty: 2-3%
Nitrate & Nitrite	AutoAnalyzer diazotization and cadmium reduction	L_D: 10 µg N/ℓ Uncertainty: 10 µg N/ℓ
pH	Submersible electrode	Uncertainty: 0.05 units
Phosphorus	AutoAnalyzer ascorbic acid reduction, persulfate digestion	L_D: 5 µg P/ℓ Uncertainty: 20% @ 10 µg P/ℓ 2% @ 60 µg P/ℓ
Primary Productivity	*In situ* C^{14}	Uncertainty: 10–20%
Silicon	AutoAnalyzer silicomolybdate complex formation	L_D: 0.03 mg Si/ℓ Uncertainty: 1%
Specific Conductance	Conductivity Meter	Uncertainty: 5–7%
Total Organic Carbon	Digestion & gas chromatographic determination of CO_2	L_D: 1 mg/ℓ Uncertainty: 10%
Temperature	Submersible Thermister	Uncertainty: 0.1°C
Transparency	Secchi disc	Uncertainty: 20%

L_D: lower limit of detection

Table 5. USEPA INTERLABORATORY COMPARISON STUDY

Species	Sample A U of M	 EPA	Sample B U of M	 EPA
Total P mg/ℓ	0.069	0.064	0.043	0.038
Soluble Reactive P mg/ℓ	0.010	0.010	0.008	0.008
$NH\text{-}_3N$ mg/ℓ	0.038	0.041	0.021	0.026
$NO_3\text{-}N$ mg/ℓ	0.268	0.252	0.631	0.610

amber bottles. BOD was measured using standard incubation techniques as outlined in Standard Methods (APHA, 1971). A Precision Instruments galvanic cell oxygen analyzer was used to measure oxygen uptake. Seeding and dilution of water samples was not necessary.

Chemical Oxygen Demand

Analysis for chemical oxygen demand (COD) was conducted on 250 mℓ water samples preserved with 0.5 mℓ concentrated sulfuric acid and refrigerated. Only a limited number of samples for White Lake were analyzed. The method of analysis used is described in Standard Methods (APHA, 1971). The samples were refluxed with 0.025 N dichromate with sulfamic acid added to prevent nitrate interference. The sample was titrated with ferrous ammonium sulfate.

Chloride

Frozen filtered water samples were used for the determination of chloride concentration. Analyses were done on the Technicon AutoAnalyzer. A colorimetric determination was used which depended on the liberation of the thiocyanate ion from mercuric thiocyanate caused by the formation of an unionized but soluble mercuric chloride compound. In the presence of ferric ions the liberated thiocyanate forms a highly colored compound, the color of which is proportional to the original chloride concentration (Technicon Instrument Corp., 1971, O'Brien, 1962, Zall *et al.*, 1956).

Dissolved Oxygen

Samples for dissolved oxygen (DO) were analyzed using the sodium azide modification of the Winkler titration (APHA, 1971). Samples were fixed immediately in the field in standard glass BOD bottles. The samples were acidified and titrated with thiosulfate aboard ship or at the end of the day in the Whitehall laboratory. The titrant was standardized daily with a biniodate solution. Additional dissolved oxygen measurements were made at 1-meter depth intervals at each lake station using a Martek Instrument Corporation submersible electrode. Measurements were calibrated using the results from the Winkler titrations.

Hydrogen Ion Activity

pH was determined using a Leeds and Northrup Model 7417 pH meter with temperature correction. A glass pH electrode with a calomel reference electrode was used. The meter was calibrated with two different pH buffer solutions each day. *In situ* measurements were also taken at each lake station at 1-meter depth intervals using a Martek Instruments Corporation submersible electrode.

Iron

Nutrient water samples were measured for iron concentrations using a wet chemical automated procedure. The method is based on the formation of a violet complex of ferrous iron with 2, 4, 6 tri (2-pyridyl) - s - triazine (TPTZ). Hydroxylamine was used to insure reduction of any trivalent iron to its divalent state and a sodium acetate buffer was used to provide the proper pH for maximum color development (Technicon Instrument Corp., 1973a). Measurements were made on a Technicon AutoAnalyzer II. Dissolved iron was measured from filtered water samples. To measure total iron, samples were first digested in a persulfate solution. The detection limit of the test was 20 μg Fe/ℓ and the uncertainty was 10 μg Fe/ℓ. In some cases, precipitation of oxidized iron compounds before analysis might have occurred due to the introduction of oxygen to anaerobic waters. This oxidation could occur during shipboard storage or pressure filtration. Consequently, peak summer hypolimnetic iron measurements are possibly low.

Ammonia Nitrogen

Ammonia concentrations were determined on the AutoAnalyzer utilizing the Berthelot reaction. In this technique a blue compound forms when a solution of an ammonium salt is added to sodium phenoxide (Technicon Instrument Corp., 1973b, USEPA, 1974). EDTA was used to prevent precipitation of hydroxides of calcium and magnesium. The accuracy of the test at low concentrations was highly sensitive to ammonia contamination in the laboratory air.

Nitrate, Nitrite Nitrogen

Nitrate was determined on the AutoAnalyzer utilizing a procedure whereby nitrate is reduced to nitrite in a cadmium column and then reacted with sulfanilamide under acid conditions to form a diazo compound. This compound then couples with N-1 napthylethylenediamine dihydrochloride to form a reddish purple azo dye, the color of which is proportional to the nitrite concentration (Technicon Instrument Corp., 1972, USEPA, 1974). Nitrite was measured separately by foregoing the reduction. All data presentations of nitrate values include nitrite.

Phosphorus

Total dissolved phosphorus was determined on filtered nutrient samples after digestion in a persulfate solution (Menzel and Corwin, 1965). The resulting released soluble reactive phosphorus was determined on the AutoAnalyzer using the Murphy and Riley (1962) ascorbic acid reduction technique (Technicon Instrument Corp., 1973c, USEPA, 1974). A phosphomolybdenum complex is formed and reduced by ascorbic acid to form a blue compound, the color of which is directly proportional to the phosphorus concentration. Total phosphorus was determined similarly except that the analysis was done on an unfiltered nutrient water sample. Measured concentrations of phosphorus from anaerobic waters may be lower than actual concentrations if oxidation of iron and manganese compounds occurs during storage, filtration, and transport.

Relative Irradiance

A submersible T.S. submarine illuminance meter equipped with two photocells and direct readout was used to measure relative light extinction. Readings were generally made at 0.5-meter intervals.

Solar Radiation

A Weather-Measure Corporation Model R401 solar radiation recorder was used to determine incident solar radiation. The unit was mounted on a hill top located next to the Whitehall laboratory.

Silicon

Dissolved silicon concentrations were determined on filtered nutrient water samples using a colorimetric AutoAnalyzer technique. This method is based on the formation of a silicomolybdate complex which is reduced by an ascorbic acid solution to form moloybdenum blue. Oxalic acid was used to prevent phosphate interference (Technicon Instrument Corp., 1973d, APHA, 1971).

Specific Conductance

Electrical conductivity was measured on alkalinity samples prior to titration using a YSI Model 31 conductivity bridge calibrated in μmho units. All measurements were corrected to 25°C. Measurements were also made *in situ* at 1-meter depth intervals at all lake stations using a Martek Instrument Corporation submersible electrode.

Total Organic Carbon

Occasional water samples were measured for total organic carbon (TOC). Samples were preserved by adding 0.5 mℓ concentrated sulfuric acid to 250 mℓ of sample. Analysis was done with an Oceanography International carbon analyzer. Samples were digested overnight in sealed ampules in a solution of potassium persulfate. The resulting carbon dioxide produced was measured.

Temperature

Water temperatures were measured using a standard mercury thermometer. *In situ* measurements were also made using a Martek Instruments Corporation submersible thermister.

Transparency

A standard Secchi disc (20 cm diameter) was used to measure water transparency. Readings were the average of the levels of disappearance and reappearance of the disc recorded in meters from the surface.

BIOLOGICAL METHODS OF ANALYSIS

Chlorophyll *a*

Samples for chlorophyll *a* analysis were collected and stored in 2-liter amber polybottles containing 5 mℓ of magnesium carbonate suspension. In the Whitehall laboratory 200 mℓ of the stored sample was filtered through a 0.45 μ Millipore filter. The filters were folded and then frozen in plastic centrifuge tubes. All storage was done in complete darkness. The samples were transported to Ann Arbor and then analyzed for chlorophyll *a* as outlined in Strickland and Parsons (1968). A Turner Model 110 Fluorometer was used for the analysis. The unit was periodically calibrated using standardized chlorophyll *a* samples and checked against spectrophotometric techniques.

Primary Productivity

Primary production was determined using an *in situ* C^{14} bicarbonate uptake technique. Two clear and one opaque 250 mℓ glass stoppered Pyrex reagent bottles were filled with water from specified depths. Each bottle was stored in the dark and then individually inoculated with two microcuries of C^{14} bicarbonate solution (Strickland and Parsons, 1968). Transfer of the inoculum from the ampules to the bottles was done with a syringe followed by rinsing. The bottles were then mounted on Plexiglas racks, lowered to their respective depths, and anchored to a station located by a separately attached buoy. Following three to four hours of incubation the bottles were retrieved and fixed with a formaldehyde solution. The contents of the bottles were later filtered onto 0.45 μ Millipore filters and rinsed with distilled water. The filters and associated suspended solids were then exposed to fuming hydrochloric acid for ten minutes to remove inorganic carbonate particles which might contain C^{14} (Wetzel, 1965). The filters were subsequently placed in 20 mℓ polyethylene liquid-scintillation vials and covered with a dioxane-based water miscible solution. The radiocarbon was determined using a Unilux I liquid scintillation counter. The carbon uptake rate was then computed considering the C^{14}, pH, and alkalinity measurements (Strickland and Parsons, 1968).

SECTION VII

GENERAL LIMNOLOGICAL OBSERVATIONS

INTRODUCTION

In any limnological study it is useful, at the outset, to provide a characterization of the lakes under investigation and to compare them to other, more intensively studied lakes. This allows both the researcher and reader to place these systems in perspective against a reference framework. In this case, the Muskegon Lakes will be compared with southern Lake Michigan (to which they discharge), to Green Bay and other areas on the Great Lakes, and among themselves. The maximum, minimum, and average values for various physical, chemical, and biological parameters measured in surface and bottom samples are presented in Table 6. In this preliminary discussion no discrimination will be made with respect to pre- and post-diversion data, the objective of this section being a general categorization of the lakes.

PHYSICAL AND CHEMICAL PARAMETERS

pH and Alkalinity

The pH of water is an expression of the molar concentration of the hydrogen ion (H^+) in solution and controls such phenomena as solubility, degree of dissociation, and acid-base equilibria. Changes in pH will occur both spatially and temporally as a result of biologic processes (CO_2 production or uptake and biodegradation) and mineral processes. The State of Michigan water quality standard for recreational waters with partial or total body contact is 6.5-8.5. The measured pH values for White Lake ranged from 7.35 to 9.06; for Muskegon Lake from 7.15 to 9.09; and for Mona Lake from 6.77 to 9.48. This range of pH as presented in Table 6 is normal for aquatic systems and represents a slightly alkaline system.

Average surface pH values were higher in Mona Lake than in White or Muskegon Lakes. This phenomenon reflects the higher rates of primary productivity in Mona Lake. Bottom water pH values were generally lower than surface values, again to a greater extent in Mona Lake. The low pH values of the bottom waters are a function of lower primary productivity and high rates of biochemical decomposition.

Alkalinity is a measure of the ability of water to neutralize acids and is an indicator of buffer capacity or the ability to resist changes in pH. A source of alkalinity is the weathering of limestone soils and

Table 6. WATER QUALITY CHARACTERISTICS—AVERAGES OF ALL DATA*

		White Lake			Muskegon Lake			Mona Lake		
Parameter (Units)	Depth	Min.	Avg.	Max.	Min.	Avg.	Max.	Min.	Avg.	Max.
pH	S	7.61	8.49	9.06	7.90	8.46	9.09	7.65	8.92	9.48
	B	7.35	7.94	8.64	7.15	7.88	8.76	6.66	8.60	8.97
Alkalinity (meq/ℓ)	S	2.04	2.67	3.44	2.10	2.74	3.89	1.79	2.26	2.96
	B	2.24	2.68	3.20	1.95	2.65	3.70	1.73	2.36	3.30
Conductivity (μmhos/cm^2)	S	271	380	460	260	329	427	331	456	566
	B	280	402	893	250	322	421	318	461	670
Chlorides (mg/ℓ)	S	13.3	35.1	53.4	9.7	20.1	27.7	18.8	36.9	49.1
	B	15.8	41.0	171.0	1.3	18.5	28.7	21.0	37.0	86.0
Dissolved Oxygen (mg/ℓ)	S	5.0	9.9	13.4	5.8	9.6	14.4	5.8	10.7	18.6
	B	0	7.5	13.1	0	7.3	14.9	0	7.8	17.9
Biochemical Oxygen Demand (mg/ℓ)	S	0.7	2.3	4.7	1.1	1.8	3.4	1.2	2.4	3.0
Chemical Oxygen Demand (mg/ℓ)	S	3	19	42	8	24	47	7	28	39
Ammonia (μg N/ℓ)	S	0	65	199	4	43	180	8	161	731
	B	6	204	1300	4	147	1514	10	775	9484
Nitrate (μg N/ℓ)	S	0	82	430	0	89	453	0	365	1189
	B	0	101	359	0	105	545	0	386	1325

Table 6.--continued

		White Lake			Muskegon Lake			Mona Lake		
Parameter (Units)	Depth	Min.	Avg.	Max.	Min.	Avg.	Max.	Min.	Avg.	Max.
Total Dissolved Phosphorus (µg P/ℓ)	S	4	14	43	4	18	74	9	107	476
	B	4	29	155	4	21	120	7	262	2180
Soluble Reactive Phosphorus (µg P/ℓ)	S	0	5	15	0	6	29	1	56	237
	B	1	17	156	0	9	99	2	166	1628
Total Phosphorus (µg P/ℓ)	S	8	35	83	17	49	316	29	185	535
	B	16	47	275	17	56	271	26	312	2411
Silicon (mg/ℓ)	S	0.09	1.75	4.67	0.05	1.43	4.16	0	1.54	4.60
	B	0.25	2.21	4.54	0.17	1.59	4.10	0	1.80	4.60
Dissolved Iron (µg/ℓ)	S	0	46	149	0	47	196	0	61	275
	B	0	48	178	0	49	401	0	79	1096
Chlorophyll *a* (µg/ℓ)	S	0.1	9.7	69.1	1.0	13.8	57.4	.3	30.7	189.8
Primary Productivity (µg C/ℓ/hr)	1m	0.1	24.8	179.7	1.0	28,5	109.6	0.1	28.2	114.7
	2m	0	12.5	89.3	0	14.9	174.6	0	7.0	62.6
Secchi Disc (m)	-	1.0	1.8	3.8	0.9	1.5	3.3	0.3	1.0	2.0

*Data are averages of all data collected between 1972-1975.

S = 2m below surface
B = 1m above bottom

rocks. Alkalinities generally ranged from 2-3 meq/ℓ for all three lakes. These levels of alkalinity are similar to those average values reported for southern Lake Michigan (2.22 meq/ℓ) and higher than those observed in the other Great Lakes (0.92-1.92 meq/ℓ) (Auer *et al.*, 1976). In this range of alkalinities the buffer capacity is high and changes in pH should be minimal.

Conductivity and Chlorides

Conductivity is a measure of the ability of water to conduct an electrical current, and, as such, reflects the ionic strength of the solution. Maximum and minimum values varied widely, but generally ranged from 270-500 μmhos/cm^2. The higher average conductivities observed for Mona Lake (surface water average of 456 μmhos/cm^2 as opposed to 380 and 329 μmhos/cm^2 in White and Muskegon Lakes) reflect more concentrated and productive conditions. These values are well above those reported for southern Lake Michigan (260 μmhos/cm^2, Auer *et al.*, 1976).

The chloride ion is a common constituent of natural waters; sources include runoff, groundwater, and wastewater discharge. Since chlorides contribute to conductivity (and both are often associated with urban and industrial contamination), it might be expected that trends in ranking among lakes would be similar; this is not entirely the case since White Lake receives chloride contamination from natural and industrial sources. Chloride values in the Muskegon Lakes (38.0 mg/ℓ for White Lake, 19.3 mg/ℓ for Muskegon Lake, and 37.0 mg/ℓ for Mona Lake) are higher than average southern Lake Michigan levels (7.2 mg/ℓ, Auer *et al.*, 1976).

Oxygen and Oxygen Demanding Substances

The maintenance of adequate levels of dissolved oxygen is important in lakes because of the effect on fish and benthic macroinvertebrate populations. Dissolved oxygen concentrations in the Muskegon Lakes vary widely, both with season and with depth. All three lakes exhibit hypolimnetic oxygen depletion in the summer months; Mona Lake to a greater extent than the others. As would be expected, dissolved oxygen concentrations above the hypolimnion are near saturation. No severe winter depletion was observed in any lake, although a reduction of 4-5 mg/ℓ was noted for the bottom water of Mona and White Lakes.

The depletion of oxygen in hypolimnetic waters is largely the result of the aerobic decomposition of organic material (oxygen demanding substances) within the water column and from the consumption of oxygen by lake sediments and the oxidation of ammonia in the nitrification process. Two general classes of substances contribute to this oxygen demand: allochthonous and autochthonous materials. The former group is composed largely of wastewater input plus organic material occurring naturally in the basin. The latter group is composed of oxygen demanding material generated within the lake, largely plankton biomass.

A study by Sands (1974) reported that the five-day biochemical oxygen demand (BOD$_5$) and chemical oxygen demand (COD) were positvely correlated with chlorophyll *a* levels (and thus phytoplankton) in the Muskegon Lakes. This suggests that autochthonous oxygen demanding materials are of primary importance in the Muskegon Lakes. The average BOD$_5$ magnitude rankings were: Mona, 2.4 mg/ℓ; White, 2.3 mg/ℓ; and Muskegon, 1.8 mg/ℓ. For COD the average values were: Mona, 28 mg/ℓ; White, 19 mg/ℓ; and Muskegon, 24 mg/ℓ; this may reflect the input of industrial waste to Muskegon Lake. The largely autochthonous nature of the oxygen demanding substances alludes to the prospect of significant improvement in dissolved oxygen levels if the quantity of phytoplankton is reduced by the wastewater diversion.

Nitrogen

The nitrogen cycle in lakes is highly complex and interfaces importantly with several facets of phytoplankton nutrition. Among the phenomena involving the nitrogen cycle observed in lakes are: fixation, nitrification, denitrification, ammonification, and uptake and assimilation of nitrogen. The transformations of nitrogen are closely linked to dissolved oxygen availability with nitrification and ammonification being aerobic processes. Nitrogen fixation and assimilation may be aerobic or anaerobic depending on the organisms involved.

Ammonia-nitrogen concentrations vary widely with depth and season in the Muskegon Lakes, but generally result in very high average levels, especially in Mona Lake. On the other hand, it should be noted that surface depletion of ammonia-nitrogen occurs in all three lakes during the summer and that high concentrations develop in the hypolimnion. Concentrations of ammonia in all White Lake waters varied from 0 to 1300 µg N/ℓ, for Muskegon Lake from 4 to 1514 µg N/ℓ, and for Mona Lake from 8 to 9484 µg N/ℓ.

Nitrate-nitrogen concentrations also reach extremely high levels in the Muskegon Lakes, especially Mona Lake. Maximum nitrate concentrations for the three lakes were 430 µg N/ℓ for White Lake, 545 µg N/ℓ for Muskegon Lake, and 1325 µg N/ℓ for Mona Lake. The presence of nitrite is largely a function of the nitrification process of which nitrite is an intermediate; when a large increase in nitrate is observed a concommitant increase in nitrite is usually noted. The nitrite/nitrate ratios vary widely, but are generally less than 1:5.

Average levels of ammonia and nitrate are 33 and 173 µg N/ℓ for southern Lake Michigan and 172 and 90 µg N/ℓ for lower Green Bay (Auer *et al.*, 1976). Mona Lake levels are 2-5 times higher than this while Muskegon and White Lake levels are generally similar. Surface values in the Muskegon Lakes, where algal uptake is important, do not always reflect the tremendous amounts of nitrogen present; bottom water values provide a better indication. The magnitude of productivity of the Muskegon

Lakes is reflected by the fact that the dissolved inorganic nitrogen concentrations, often observed to approach or exceed 1 mg N/ℓ in the winter, are depleted near zero in the summer months, revealing a tremendous potential production of phytoplankton. The surface water depletion of dissolved inorganic nitrogen in White Lake and Muskegon Lake encourage the dominance of the phytoplankton by nitrogen-fixing blue-green algae such as *Anabaena* sp., and *Aphanizomenon Flos-aquans* (USEPA, 1975a,c, Auer and Canale, 1976).

Phosphorus

Phosphorus, like nitrogen, is an important plant nutrient. The dynamics of phosphorus in aquatic systems revolve largely around phytoplankton uptake with subsequent decomposition and release and sediment-water interchange. The former process is to a large extent seasonal and the latter is closely tied to the oxygen status of the hypolimnion.

Three forms of phosphorus were measured in this study: total phosphorus, total dissolved phosphorus, and soluble reactive phosphorus. Average total phosphorus concentrations in Muskegon and White Lakes are 2-3 times greater than those reported for southern Lake Michigan (14.6 μg P/ℓ) but less than those observed for lower Green Bay (304 μg P/ℓ). Mona Lake average total phosphorus levels (248 μg P/ℓ) are an order of magnitude greater than those reported for southern Lake Michigan. The release of large quantities of dissolved phosphorus to the bottom waters of all three lakes from sediments and phytoplankton settling and decomposition is evident from the data. The bottom water total dissolved phosphorus maximum in Mona Lake (2411 μg P/ℓ) was an order of magnitude greater than that for either Muskegon Lake (271 μg P/ℓ) or White Lake (275 μg P/ℓ). Additionally, the 2m average total dissolved phosphorus concentration for Mona Lake exceeds that of the particulate fraction, the only lake in which this happens. This observation may indicate that dissolved phosphorus levels in Mona Lake are at times in excess of that which can be utilized by the organisms. Two-meter total dissolved phosphorus is depleted to near the analytical limit of detection at times in all lakes, but never to zero.

Silicon

Silicon is an element required for growth by a number of aquatic organisms, most notably the diatoms (Bacillariophyceae). Silicon, as silicon dioxide, generally occurs abundantly in natural waters and is seldom introduced in high levels as a result of man-generated pollution. Since this element is required by diatoms, its availability serves to control species succession in the phytoplankton. Silicon levels in the Muskegon Lakes are, on the average, quite high (1.4-1.7 mg Si/ℓ). The silicon depletion observed in all three lakes reflects the large diatom populations observed at certain times of the year. Average dissolved silicon values for the Muskegon Lakes are approximately 3 times those observed for southern Lake Michigan (0.43 mg/ℓ) and similar to those observed for lower Green Bay (Auer *et al.*, 1976).

Iron

Iron is also an element required, in small quantities, for algal growth. Iron participates in a number of complex chemical reactions in aquatic systems, often those associated with changes in oxygen concentrations and redox potential. Large differences were noted between maximum and minimum levels of iron in the Muskegon Lakes; these are the result of precipitation and dissolution reactions. Under aerobic conditions, ferric iron undergoes reactions involving phosphorus and hydroxyl ions which result in the formation of ferric compounds existing in the solid phase. Some of this material remains suspended as colloidal particles in the water, while the remainder becomes part of the sediment. Additionally, iron is taken up by the phytoplankton and transferred to the sediment upon the death and settling of the organisms. During stratification, under anaerobic conditions, the ferric iron is reduced to the more soluble, ferrous form. At this point the iron is released to the water column and continues the cycle. Dissolved iron concentrations are an order of magnitude higher in the Muskegon Lakes than in Lake Michigan (6 μg/ℓ) (Auer *et al.*, 1976).

BIOLOGICAL PARAMETERS

Chlorophyll *a*

Chlorophyll *a* concentrations, although variable with the physiological state of the organism, provide a measure of the standing stock of phytoplankton present in a lake at any point in time. High levels of chlorophyll *a*, in excess of 20 μg/ℓ especially when contributed by blue-green algae (Cyanophyceae), may lead to reduction in water quality for many uses. The average values for the Muskegon Lakes are 5-15 times those observed for southern Lake Michigan and bracket that of lower Green Bay (Auer *et al.*, 1976), with Mona Lake averaging 2-3 times more pigment than the other lakes at 30 μg/ℓ. Concentrations were observed as high as 69 μg/ℓ in White Lake, 57 μg/ℓ in Muskegon Lake, and 189 μg/ℓ in Mona Lake.

Primary Productivity

Primary productivity, as measured by C^{14} uptake, is an indication of the phytoplankton growth rate actually occurring at a specific depth and time. As such, this uptake is a function of specific conditions of light, nutrient availability, and the standing stock of phytoplankton. Again these values are 5-10 times greater than those measured for southern Lake Michigan (Auer *et al.*, 1976). The distinct range observed for chlorophyll *a* levels (ranking the lakes by greatest average concentration: Mona, Muskegon, White) is not observed for primary productivity. The reason for this is the presence of a self-shading feedback mechanism which reduces the average primary productivity in lakes with extremely large standing stocks of phytoplankton. Maximum primary productivity measurements were 179 μg C/ℓ/hr for White Lake, 174μg C/ℓ/hr for Muskegon Lake, and 144 μg C/ℓ/hr for Mona Lake.

Secchi Disc

The depth at which the Secchi disc is visible is a measure of water transparency. It is influenced by changes in phytoplankton standing crop, suspended solids, and other turbidity components which scatter, reflect, and absorb light. The trend in Secchi disc among lakes is identical to that observed for chlorophyll *a*. Average Secchi disc measurements were 1.82 for White Lake, 1.54 for Muskegon Lake, and 1.01 for Mona Lake. By way of comparison, southern Lake Michigan Secchi disc values average 5.2 m (Auer *et al.*, 1976).

Phytoplankton, Zooplankton, and Benthos

Data on biologic taxonomy were available from studies by the USEPA (1975a, 1975b, 1975c, 1975d). The phytoplankton of the Muskegon Lakes is dominated by eutrophic forms of blue-green algae (Cyanophyceae) and diatoms (Bacillariophyceae). Nitrogen-fixing blue-greens and pennate diatoms are found largely in Muskegon and White Lakes while non-nitrogen-fixing blue-greens (primarily *Microcystis aeruginosa*) and centric diatoms are found in Mona Lake. The zooplankton of all three lakes is dominated by *Bosmina longirostris*, a eutrophic form. Numbers of zooplankton decrease in abundance among Mona and Muskegon and Muskegon and White Lakes. The dominant benthic macroinvertebrate in Mona Lake is the phantom midge, *Chaoborus*. Oligochaetes dominate the bottom fauna of Muskegon and White Lakes.

SUMMARY

In summary it can be said that the Muskegon Lakes are highly productive lakes, with ample phytoplankton growth nutrients (N, P, Si) to permit massive algal blooms. That such events occur is evidenced in surface water nutrient depletion and extremely high levels of chlorophyll *a* and primary productivity with concommitant reduction in Secchi disc depth and hypolimnetic dissolved oxygen concentrations.

SECTION VIII

WHITE LAKE

INTRODUCTION

The discussion of the effect of the wastewater diversion project on the water quality of the Muskegon Lakes will be presented in two major parts. The first of these will involve tributary-related considerations, e.g., hydrology, contaminant concentrations and loading. The evaluation of the response of water quality in White, Muskegon, and Mona Lakes to the wastewater spray-irrigation project, requires information on the past and present pollutant loads to the system. Contaminant loads from direct municipal and industrial discharges and from tributary and non-point sources must be quantified separately. The changes in pollutant loads to the lakes and the subsequent response of lake water quality can, in this manner, be properly related to the implementation of the diversion project. Information is also needed regarding the hydrology of the system. Hydrological characteristics dictate the rate at which water, nutrients, and algae are flushed from the lakes.

The second part of the discussion involves lake-related considerations, e.g., spatial, seasonal, and long-term trends in lake water quality. This section of the report will deal with lake chemistry and biology, primarily focusing on major algal growth nutrients (N, P, Si) and phytoplankton indicators, since these materials are those expected to change as a result of the diversion program.

An evaluation of spatial and temporal trends in major phytoplankton growth nutrients is fundamental to an understanding of variation in plankton populations. In order to determine if changes in water quality are a result of the diversion, their relation to the annual cycles in the lake must be studied. Since the vertical distribution of chemical and biological species are closely related to seasonal phenomena, both spatial and seasonal characteristics will be discussed together. Long-range trends and their relation to the wastewater diversion project will be analyzed subsequently. A brief discussion of the biological characteristics of each lake and summaries of specialty studies (where applicable) are presented.

White Lake is the northernmost of the three study lakes. It is approximately 7 kilometers long and 0.7 to 1.0 km wide with a southwest to northeast orientation (see Figure 7). The lake receives flow from nine tributaries, dominanted by the White River. The lake empties into Lake Michigan through a narrow channel. The channel is dredged and is navigable, qualifying White Lake as a shelter for ship refuge.

The lake has a surface area of 10.4 km^2, a volume of 7.6 x 10^7 m^3, and a mean depth of 7.3 meters. The lake is naturally divided into three basins: east, central, and west. The central basin is the deepest with a maximum depth of approximately 23 meters. The lake has extensive shallow areas which support prolific macrophyte growth. Figure 4 describes lake morphology.

The region surrounding White Lake is principally residential. The lake received discharges from three major industries prior to diversion. (Hooker Chemical Company, Misco Division of Howmett Corporation, and the Whitehall Leather Company). Two of these industries have not yet fully diverted into the system. One municipal wastewater treatment plant (Whitehall) discharged to White Lake prior to diversion.

TRIBUTARY-RELATED CONSIDERATIONS

Hydrology--The White Lake drainage basin contains 1,319 km^2 of land in Muskegon, Newaygo, and Oceana Counties. The White River drainage area accounts for 94% (1,245 km^2) of the total lake drainage basin (see Table 7). The remaining area includes minor lake tributaries and direct drainage to the lake. The White River contributes an important share of the flow to White Lake. This is seen in hydrologic data collected by the Michigan Water Resource Commission in 1967 (see Table 8) (MWRC, 1967). This publication reports that the White River accounts for approximately 99% of the total flow to White Lake.

The White River begins in north central Newaygo County and flows southwest to the mouth. Although technically direct drainage to White Lake is considered part of the river basin, in this report, discussion of the White River will only include the drainage up to its entry into White Lake. The White River basin is primarily rural with approximately 40% of the land forested and 39% agricultural (MWRC, 1967). The flow in the White River has been monitored by the USGS since 1958. The 17-year average flow at this location is 11.4 m^3/sec. However, the annual average flow varies significantly (see Figure 8). During the period of this study the annual flow was observed to vary. During the first year (1972) a low flow was recorded, a moderate flow was measured in 1973, and high flows in 1974 and 1975. The average monthly flows measured during the investigation are presented in Figure 9. These data reveal that the seasonal effects of runoff exert a significant influence on the White River flow.

The flow as measured at the USGS station does not represent the total flow of the White River into White Lake. Between the USGS station and White Lake the river receives additional direct drainage from 11 tributaries. Flow at the river mouth was estimated by multiplying the flow at the gauging station by a factor of 1.4. The factor was obtained by comparing USGS measured flows to flows measured at the mouth during this study.

Table 7. WHITE LAKE DRAINAGE BASIN AREAS

Description	Area (km^2)	Percentage of Total
White River	1245.1	94.4%
Pierson Dam	27.0	2.0%
Mill Pond Creek	21.4	1.6%
Miscellaneous Tributaries & Direct Drainage	15.4	1.2%
Lake Surface Area	10.4	0.8%
Total	1319.3	100.0%

Table 8. WHITE LAKE TRIBUTARY FLOW DATA, JULY 11 AND 12, 1967 (MWRC, 1967)

Name	Discharge (m^3/sec)	Percentage of Total
Unnamed Creek from City Solid Waste Disposal Site	0.018	0.2%
Unnamed Creek	0.0004	trace
Mill Pond Creek	0.079	0.6%
Wildcat Creek	0.021	0.2%
Birch Creek	0.027	0.2%
Strawberry Creek	0.010	trace
Pierson Drain	0.026	0.2%
Coon Creek	0.007	trace
White River	13.28	98.6%

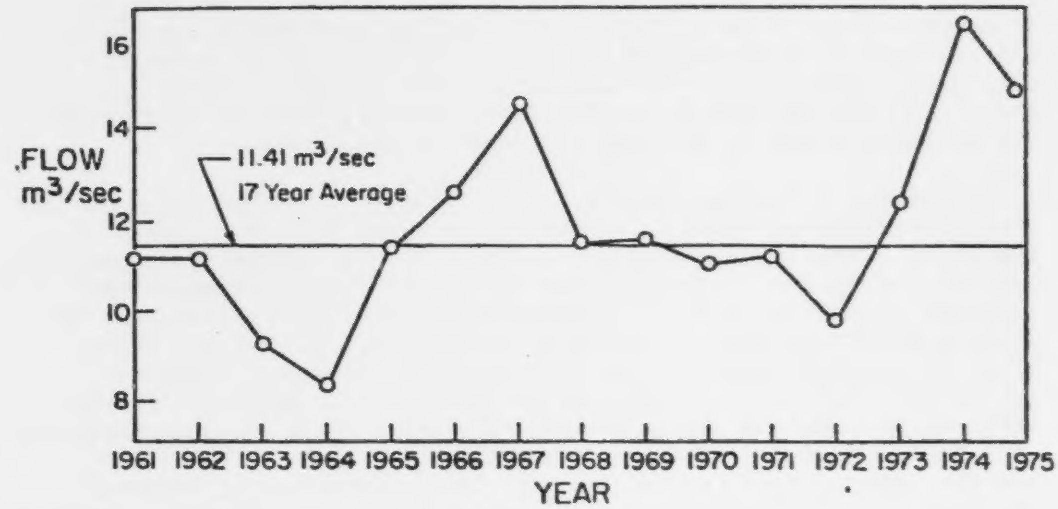


Figure 8. White River average annual flow at Whitehall, Michigan (USGS Station # 04122200).

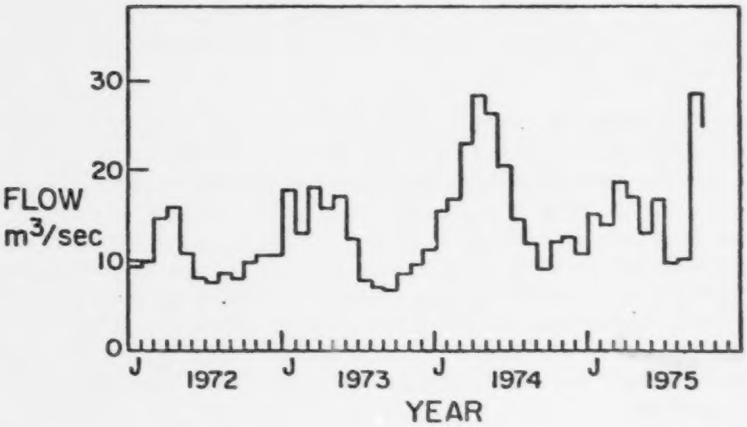


Figure 9. Mean monthly flow of the White River at Whitehall, Michigan (USGS Station # 04122200).

Direct municipal and industrial discharges do not significantly affect the hydrologic characteristics of White Lake.

The significance of the river flow to White Lake can be estimated if the ratio of lake volume to tributary flow, or mean hydraulic retention time, is calculated. Dividing the total lake volume of 7.6×10^7 cubic meters by the total average flow, the retention time in the lake is calculated to be 55 days. On the average, the flow coming into White Lake is capable of exchanging the total lake water volume seven times each year. This calculation does not account for short-circuiting, stratification, and stagnation in littoral zones, nor does it describe the seasonal variation of river flow and retention time.

Concentration of Chemical Species in Tributaries -- Various water quality parameters were monitored in the White River and Mill Pond Creek. A summary of these results is given in Table 9. The average concentrations measured in the two tributaries were of the same order of magnitude although certain White River parameters had higher concentrations. The three-year average concentrations of chemical species for the White River at the mouth were 17.6 mg Cl/ℓ chloride, 60 μg N/ℓ ammonia, 136 μg N/ℓ nitrate, 46 μg P/ℓ total phosphorus, 2.8 mg Si/ℓ dissolved reactive silicon, and 223 μg/ℓ iron. The Mill Pond Creek concentrations were half as high for chlorides, approximately equal for nitrate and ammonia, one-third to one-half as high for phosphorus, approximately the same for silicon and slightly lower for iron. Because Mill Pond Creek has such a small flow relative to the White River and its concentrations are generally lower than those in the White River, it is not considered to have a significant influence on White Lake water quality. This is assumed true for other lake tributaries as well. These streams have drainage basins of similar character to that of Mill Pond Creek and receive no direct wastewater discharges. This similarity in water quality was verified by a Michigan Water Resources Commission (1967) study. All future discussion of river water quality will be focused on the White River.

Water quality in White Lake, particularly subsequent to the wastewater diversion, is affected by the tributary water quality. The White River upstream of the influence of the diversion and spray-irrigation project at station 201 is rich in algal nutrients. Monitored total phosphorus concentrations of approximately 46 μg P/ℓ, dissolved inorganic nitrogen concentrations of 244 μg N/ℓ, and dissolved silicon concentrations of 3.3 mg Si/ℓ are sufficient to support algal growth at nuisance levels. Although only about one-half of the total phosphorus is in the soluble reactive form, and considered immediately available for algal growth, all of the phosphorus is potentially available through bacterial and enzymatic action.

Table 9. AVERAGE CONCENTRATIONS OF SELECTED CHEMICAL SPECIES IN TRIBUTARIES TO WHITE LAKE, 1972-1975

Tributary	Year	Chloride (mg Cl/ℓ)	Ammonia (μg N/ℓ)	Nitrate (μg N/ℓ)	Soluble Reactive Phosphorus (μg P/ℓ)	Total Dissolved Phosphorus (μg P/ℓ)	Total Phosphorus (μg P/ℓ)	Silicon (mg Si/ℓ)	Dissolved Iron (μg Fe/ℓ)	Total Iron (μg Fe/ℓ)
White River (at mouth)	1973	19.1	45	118	16	29	51	2.9	104	368
(Station 227)	1974	17.2	52	142	14	25	42	2.9	87	179
	1975	16.7	87	147	15	27	45	2.7	88	162
	1973-1975	17.6	60	136	15	27	46	2.8	92	223
White River	1972	18.5	86	260	--	16	26	4.1	--	--
(Station 201)	1973	18.3	37	108	11	22	46	3.1	113	489
	1974	17.8	32	133	16	25	37	3.3	95	154
	1975	16.2	37	171	13	25	46	3.0	68	199
	1972-1975	17.8	43	155	14	23	40	3.3	91	261
White River	1972	8.9	56	335	--	21	28	3.9	—	--
(Station 202)	1973	7.2	23	167	28	31	49	3.2	58	342
	1974	7.3	31	182	29	41	51	3.2	76	123
	1975	7.2	31	217	23	36	49	3.1	68	144
	1972-1975	7.5	33	211	27	33	46	3.3	68	190
Mill Pond Creek	1973	10.6	45	107	6	16	25	2.6	58	265
(Station 226)	1974	9.7	57	183	9	14	24	2.7	64	123
	1975	7.0	38	177	9	16	28	2.9	48	104
	1973-1975	9.5	49	154	8	15	25	2.7	59	162

Figures 10 to 19 show plots of the concentrations of various parameters measured in the White River. Total phosphorus concentrations were low (approximately 20 μg P/ℓ) during periods of low flow and runoff (fall and winter) when the river water originates principally from groundwater or snow melt. Groundwater is generally low in phosphorus due to soil adsorption. Beginning in the spring concentrations of phosphorus increase to levels approaching 100 μg P/ℓ. This elevation is a consequence of runoff from forest and agricultural lands carrying nutrient rich soil and other allochthonous materials into the river. Nitrate concentrations were observed to be highest during the winter months at times exceeding 500 μg N/ℓ and lowest during the summer (approximately 50 μg N/ℓ). Ammonia concentrations show less variation than phosphorus or nitrate but appear to be lower in the spring. Concentrations averaged about 50 μg N/ℓ. The nitrate increase in the winter is probably the consequence of low uptake by plants and the influx of nitrate rich groundwaters. Lower spring ammonia concentrations may reflect dilution by high spring runoff. Nitrate and ammonia concentrations decrease simultaneously in the spring.

Comparisons of the yearly average concentrations of chemical species at the mouth of the White River (Station 227) before and after the wastewater diversion reveal only a few perceptible changes. Iron concentrations were noted to increase during 1975; and nitrate concentrations were observed to increase subsequent to 1973. Other parameters such as phosphorus, silicon, and chloride concentrations were not observed to change significantly. It is difficult to determine if the observed changes in concentration were a consequence of the sewage diversion spray-irrigation project because of the lack of sufficient data. No observations were made at the river mouth (Station 227) during 1972. Sample data were available for 1972 from Station 202 located upstream of the former Whitehall Wastewater Treatment Plant. Only the 1975 data reflect full implementation of the spray-irrigation project. Attempting to draw conclusions regarding changes in nutrient concentrations resulting from the diversion project at Station 227 can be misleading because short-term alterations in upstream concentration may be important. Comparison of upstream and downstream concentrations in the White River is therefore important.

Stations 201 and 202 are located upstream and downstream, respectively, of the spray-irrigation site and Stations 202 and 227 are located upstream and downstream, respectively, of the former Whitehall Wastewater Treatment Plant discharge (see Figures 7 and 20). An evaluation of the effects of the spray-irrigation drainage and non-point sources can be made by comparing concentrations among these stations.

Significant changes in phosphorus concentrations were not observed spatially or temporally during the period of study. This is true with

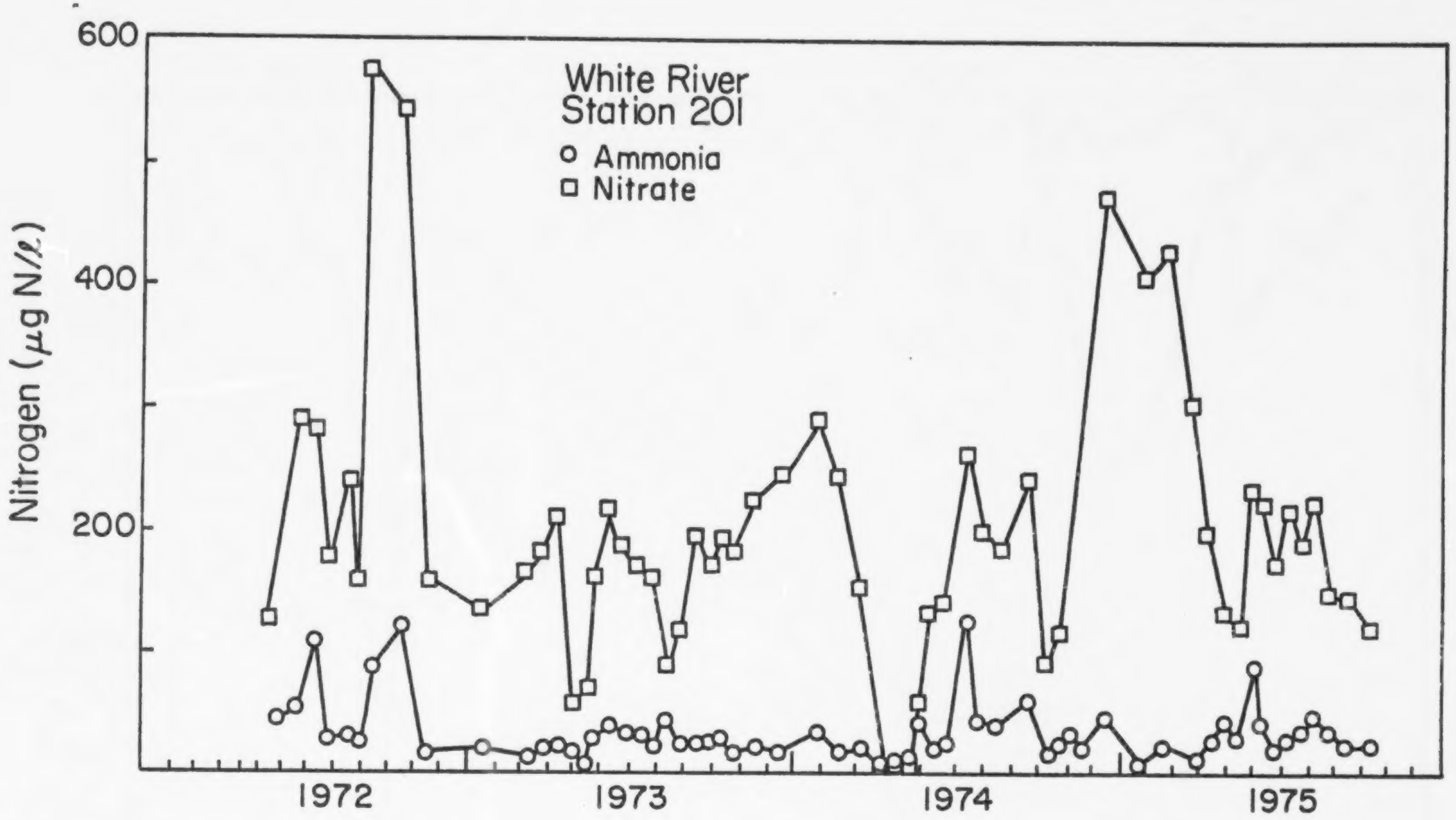


Figure 10. Dissolved inorganic nitrogen concentrations at Station 201 in White River, 1972-1975.

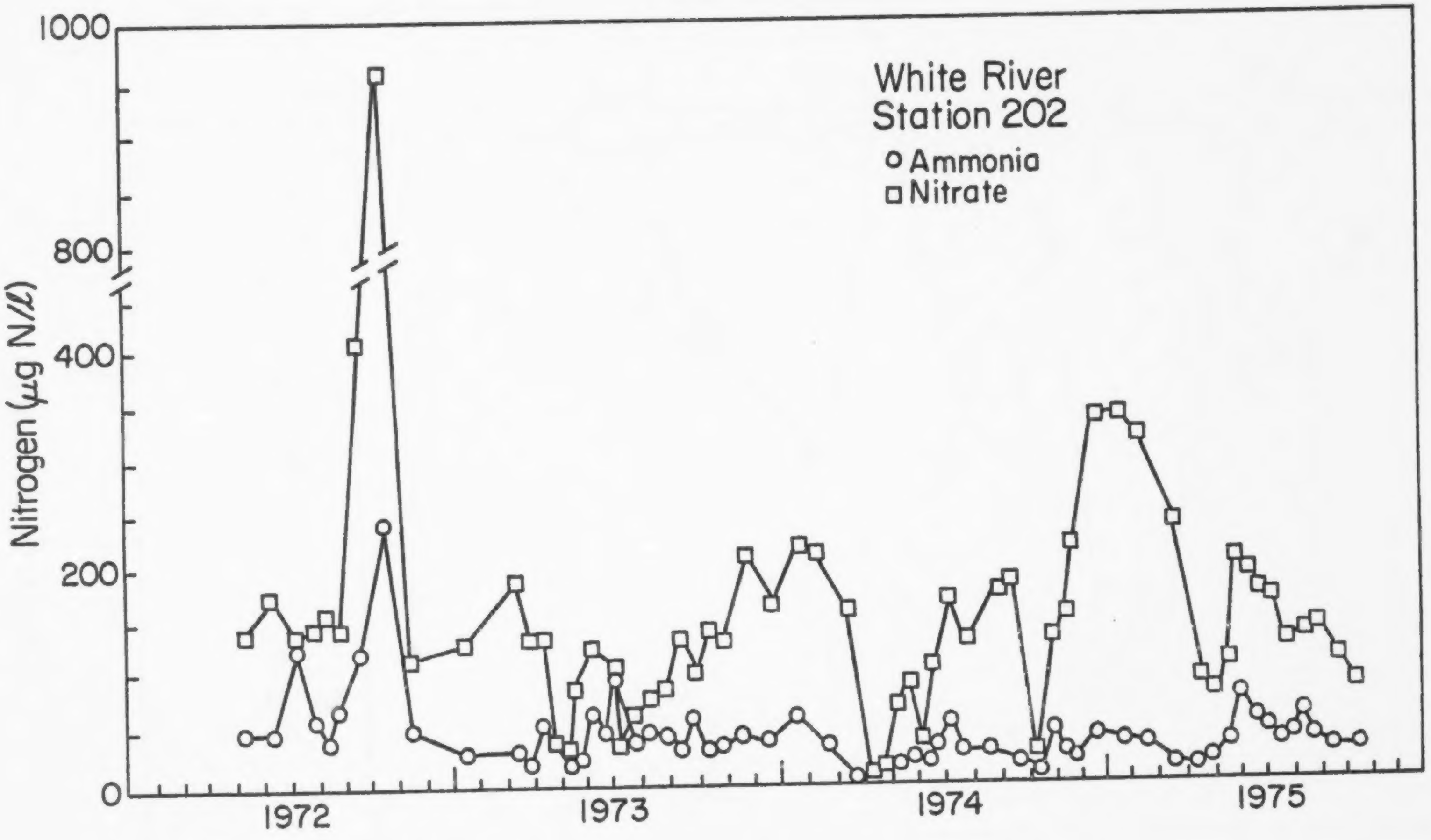


Figure 11. Dissolved inorganic nitrogen concentrations at Station 202 in White River, 1972-1975.

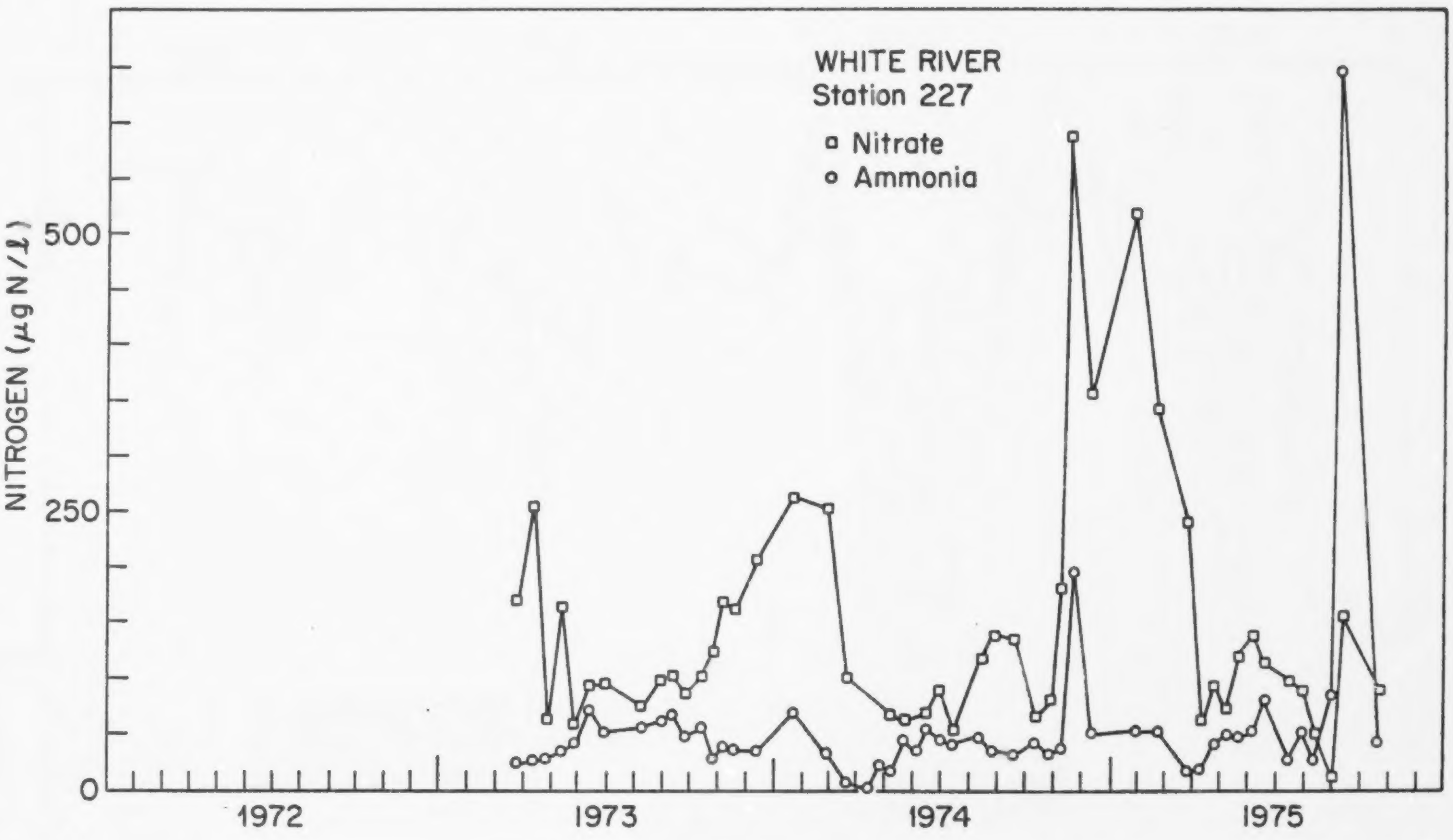


Figure 12. Dissolved inorganic nitrogen concentrations at Station 227 in White River 1973-1975.

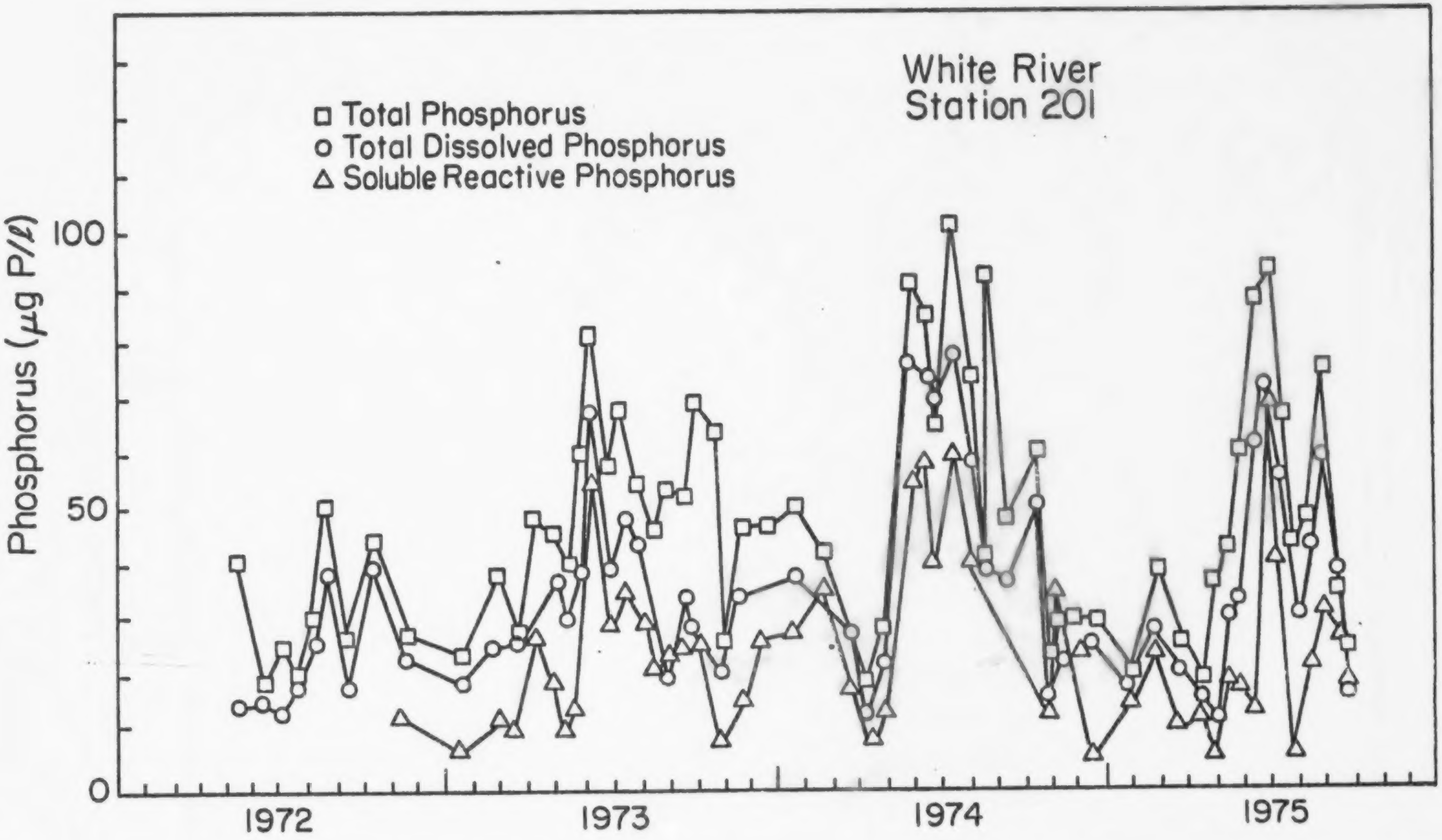


Figure 13. Phosphorus concentrations at Station 201 in White River, 1972-1975.

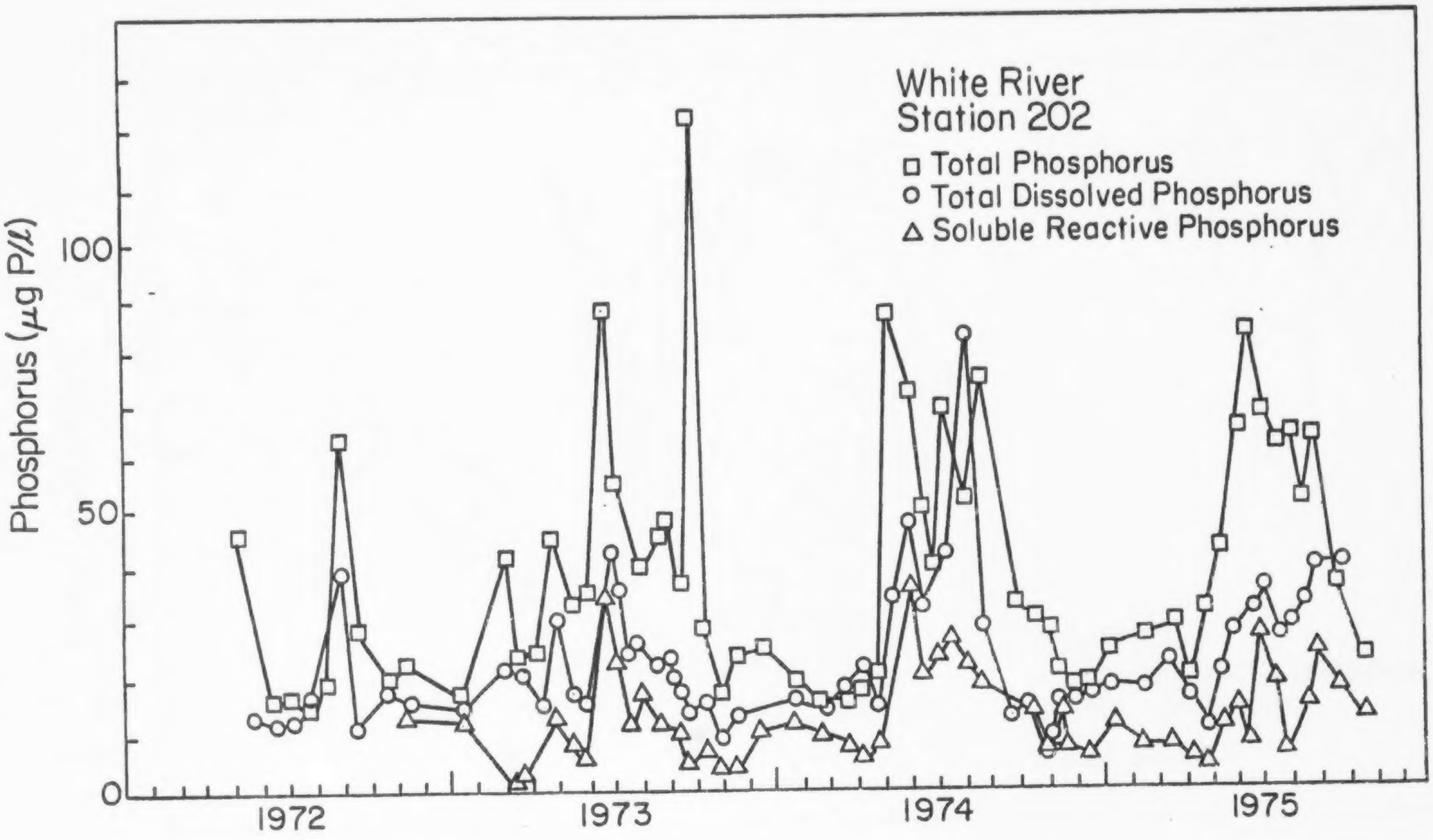


Figure 14. Phosphorus concentrations at Station 202 in White River, 1972-1975.

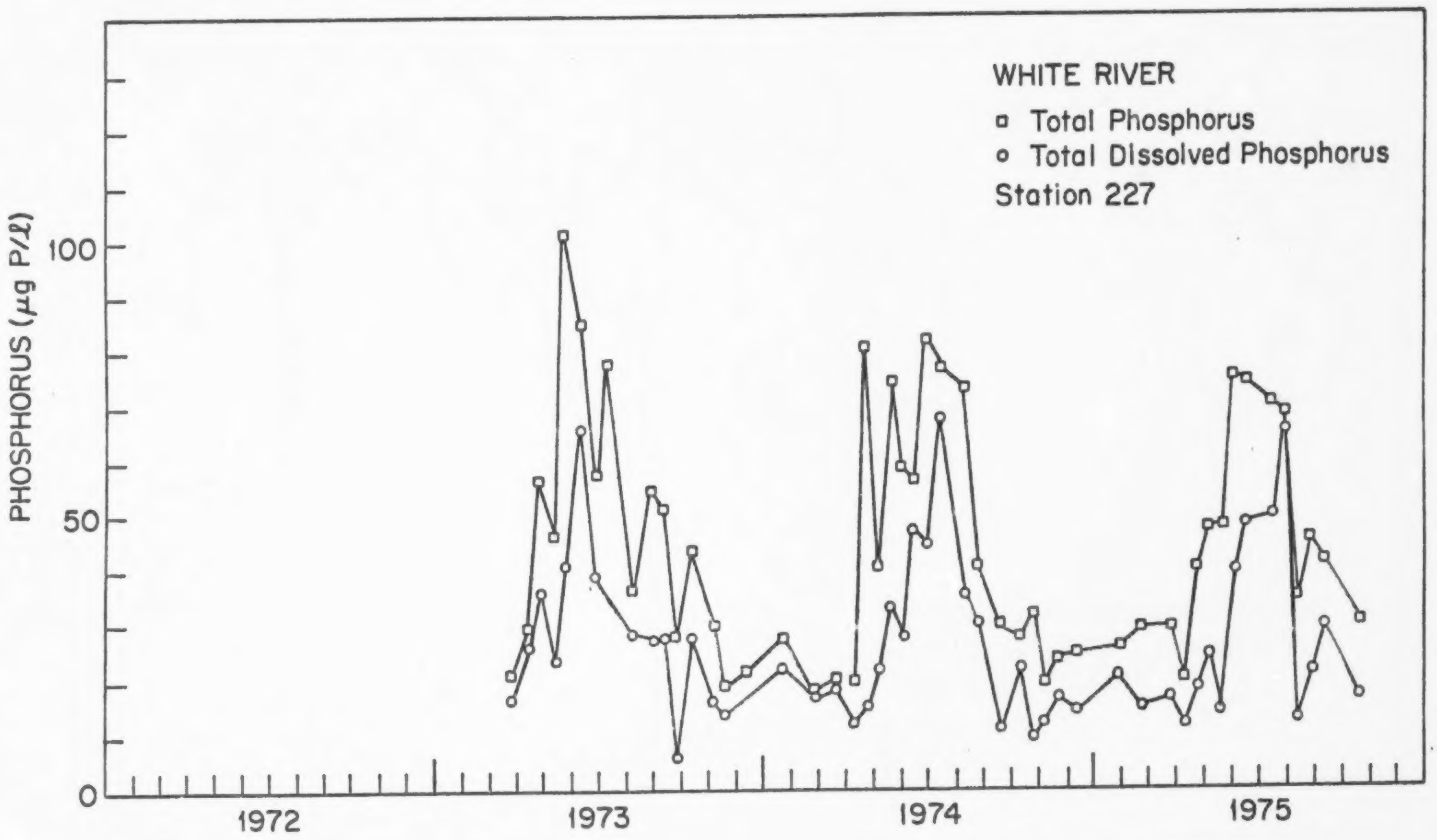


Figure 15. Phosphorus concentrations at Station 227 in White River, 1973-1975.

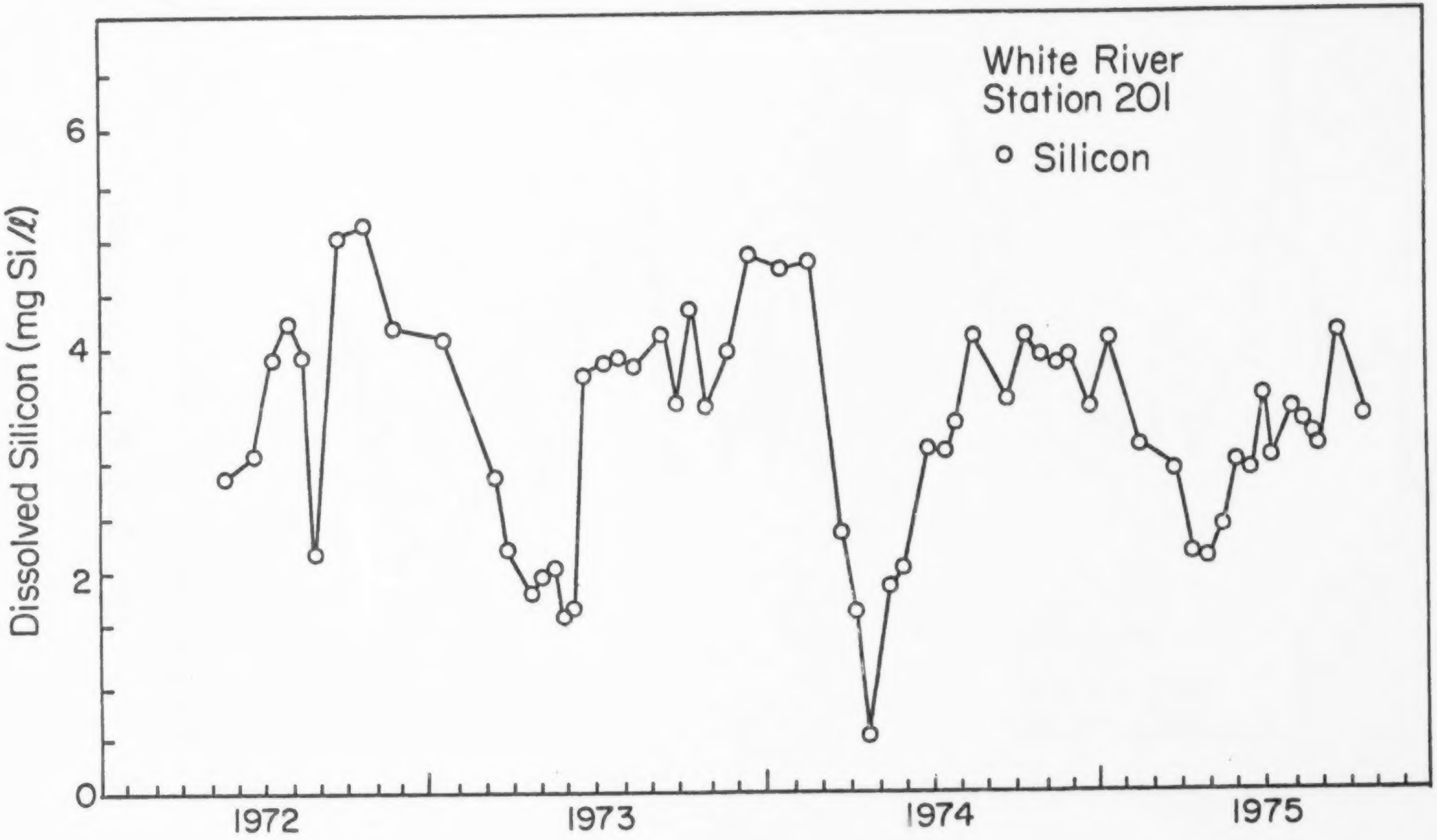


Figure 16. Silicon concentrations at Station 201 in White River, 1972-1975.

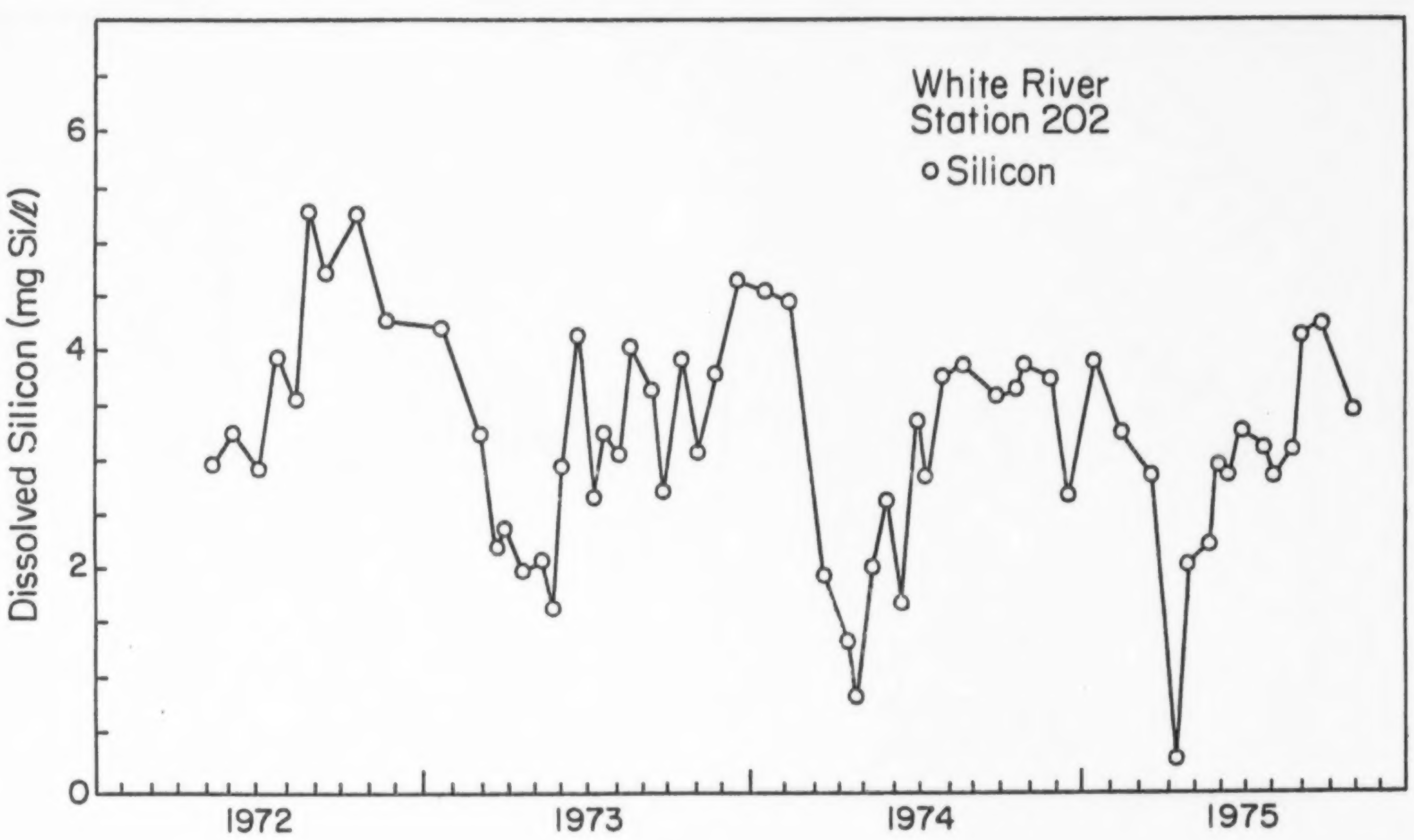


Figure 17. Silicon concentrations at Station 202 in White River, 1972-1975.

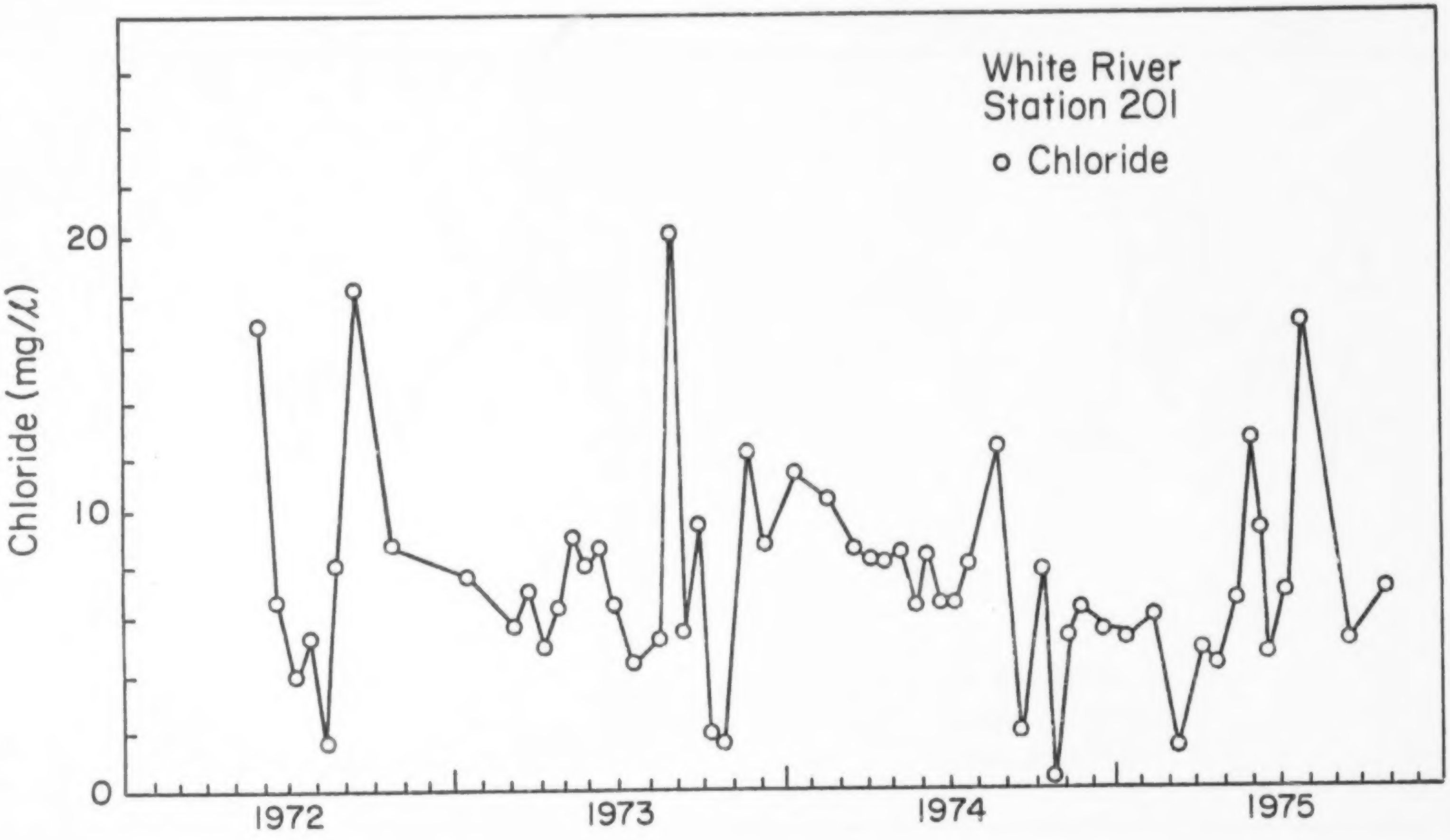


Figure 18. Chloride concentrations at Station 201 in White River, 1972-1975.

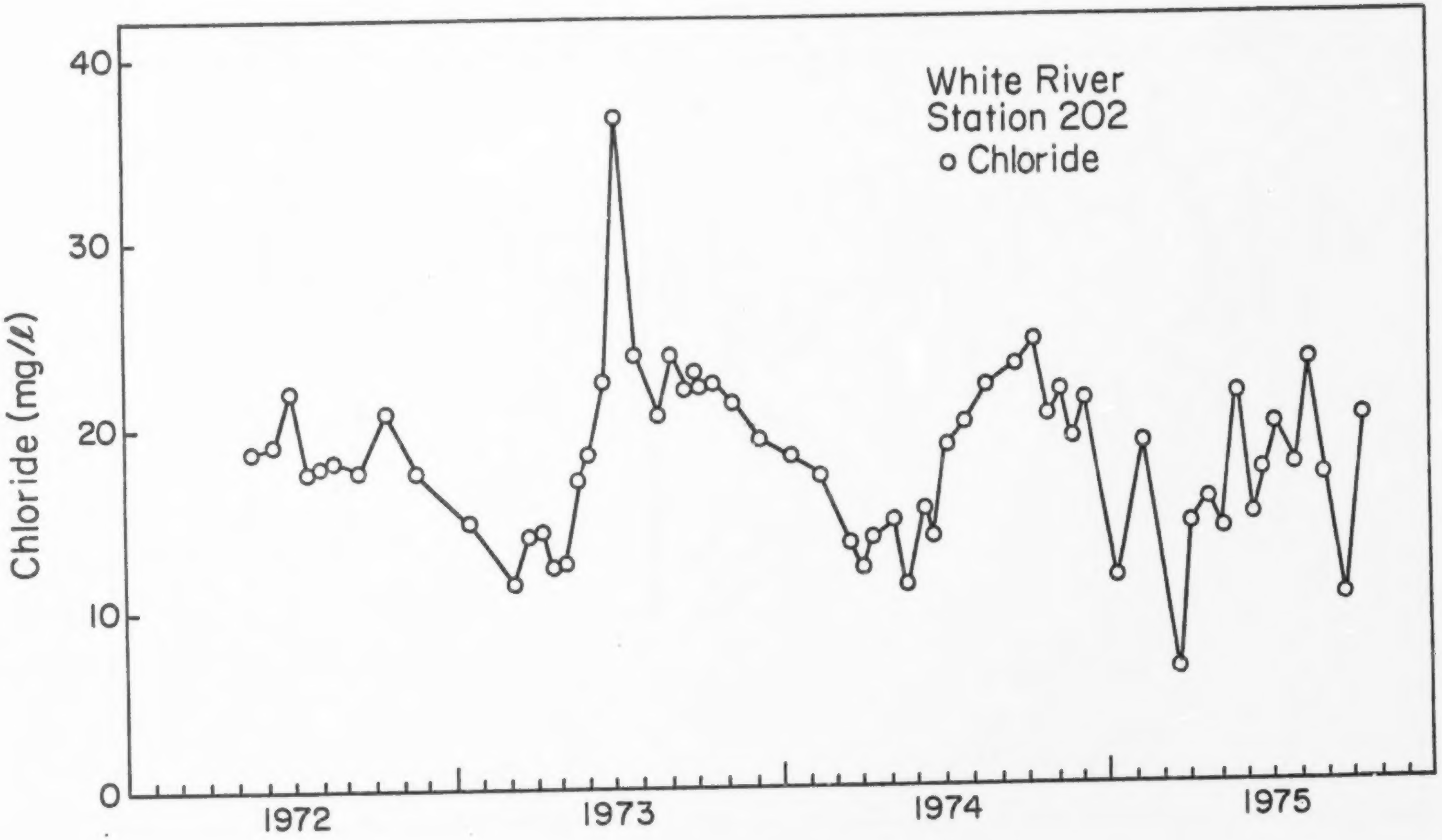


Figure 19. Chloride concentrations at Station 202 in White River, 1972-1975.

respect to all forms of phosphorus measured. During the period of 1973 prior to diversion slightly higher phosphorus concentrations were observed at times at the rivermouth as compared to the upstream stations. Consistent trends were not observed, however, and statistical support for this observation cannot be demonstrated due to limited data. In general, it appears that upstream non-point sources are the main factor determining phosphorus concentrations in the White River. No increase in phosphorus levels in White River were observed to result from the spray-irrigation drainage. Only limited effects on phosphorus concentrations can be attributed to the former Whitehall Wastewater Treatment Plant discharge. Had more data been available from the river mouth (Station 227) prior to diversion (1972 and earlier), the latter effect may have been more noticeable.

Spray drainage is discharged from the Whitehall spray-irrigation site to the White River after ground infiltration. Concentration in wells surrounding the area then characterize the spray drainage. Measurements from these wells reveal concentrations similar to those from the upstream White River (Appendix E). Average total phosphorus concentrations were approximately 50 μg P/ℓ; ammonia was 200 μg N/ℓ in 1974 and 50 μg N/ℓ in 1975; and nitrate-nitrite was approximately 60 to 70 μg N/ℓ. As a consequence, no significant concentration changes were observed to result from the spray drainage in the White River.

Significant spatial and temporal variations in nitrate concentrations were observed. Ammonia concentrations were also observed to fluctuate. Analysis of these data is complicated by the processes of nitrification, denitrification, ammonification, and algal uptake. Also complete mass balance considerations cannot be conducted without data on total nitrogen. It appears, however, that the nitrate concentrations are highest at the upstream Station (201); no changes in nitrate were observed to result from nutrient (fertilizer) runoff at the spray-irrigation site. The high upstream nitrate is possibly groundwater derived. Ammonia concentrations increased downstream. The largest increases in ammonia were observed to occur between Stations 202 and 227. This occurrence is considered to be an effect of non-point loadings, decomposition, and ammonification of organics in the river. It appears that unidentified non-point sources of inorganic nitrogen possess sufficient inherent variation to mask the effects of known point sources.

Silicon concentrations were not observed to vary temporally but did appear to decrease spatially between Stations 202 and 227. This might be a consequence of dilution by lower basin tributaries. The influence of White Lake on the integrity of lower White River waters must also be considered. Because of the dominant westerly winds, the lake at times has a tendency to mix with White River waters and alter their normal character. This problem may exist not only for silicon but also for all concentration measured at the river mouth station (227). Hence, the absence of any strong measurable effect from the wastewater treatment plant discharge could have been partially a consequence of the close proximity of Station 227 to the lake. (Logistics and geographical restrictions prevented the designation of a station further upstream, yet still downstream of the former wastewater treatment plant.)

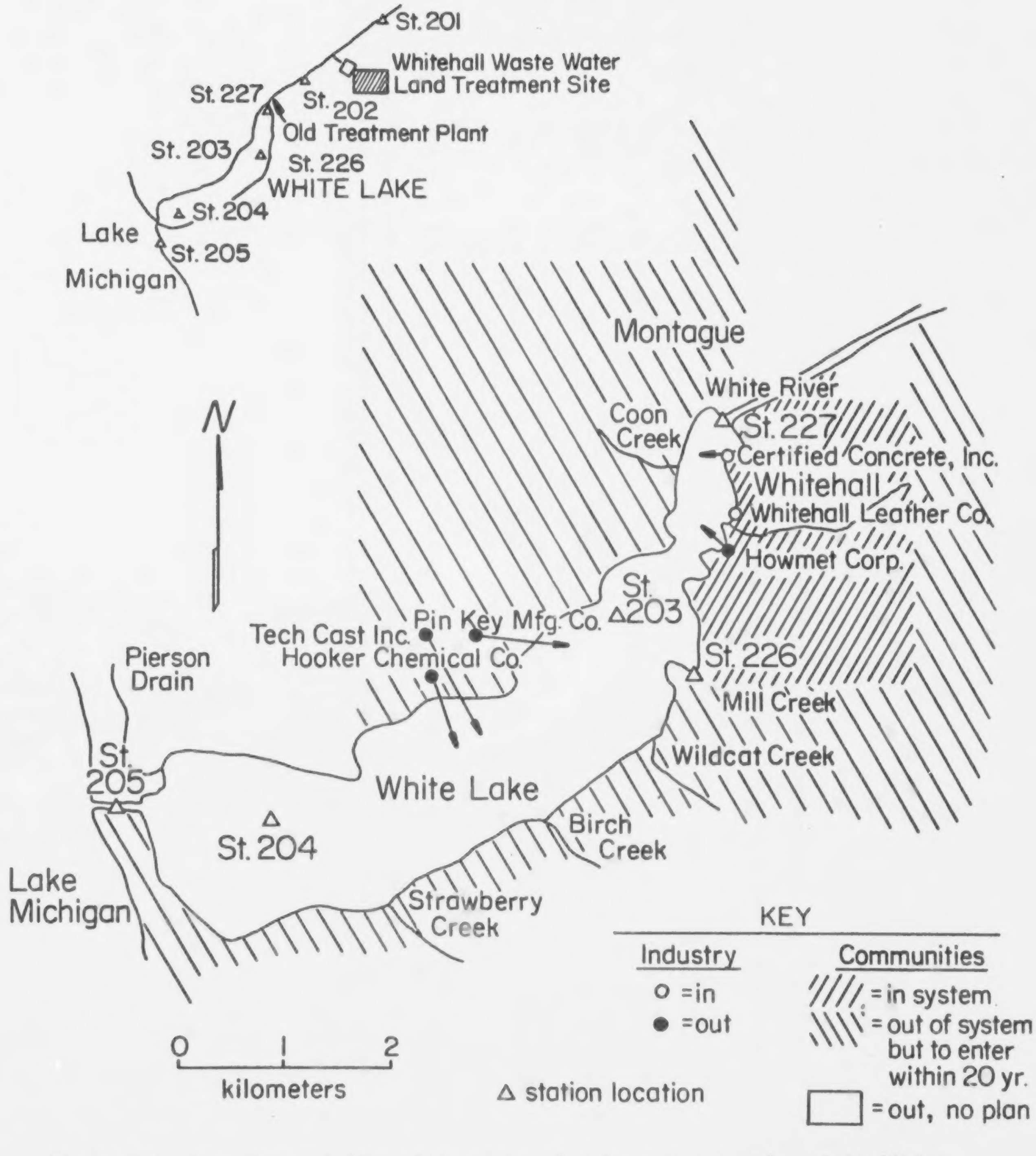
Chloride concentrations were not observed to change year to year, but a large increase in concentration was observed spatially between Stations 201 and 202. This increase also was noted prior to the spraying period. Because no known discharges of chloride exist in this portion of the basin, the increase may be an effect of urban runoff or groundwater infiltration. The Whitehall area is known to lie over brine deposits and local chemical companies rely on these deposits for their operation.

Concentrations of iron were observed to vary between stations and among years. Higher concentrations of iron were measured in 1973. No explanation is proposed for this observation, except variation in unidentified non-point contamination and the relative influence of groundwater high in iron.

No significant alterations in White River chemistry could be observed to result from the diversion spray-irrigation project because the spray discharge has concentrations similar to those in the upper river. Furthermore, occasional influences of White Lake water on tributary samples at the lowermost White River Station (227), variability in upstream non-point loads, and limited pre-diversion river mouth data prevent detailed statistical analysis of the data. Calculations comparing non-point and point nutrient loads are required to evaluate the effectiveness of the system. This analysis will follow.

Nutrient Loads -- Nutrient loads to White Lake originate from tributary, industrial, and municipal sources. Of the nine tributaries discharging to White Lake, the White River is the dominant hydrologic and nutrient source. In past studies on White Lake tributaries, the White River was estimated to contribute 98% of the total phosphorus loading and 96% of the total nitrogen load (MWRC, 1967). The focus of this investigation of tributary monitoring and load calculations was therefore on the White River with minor emphasis on Mill Pond Creek, the next largest hydrologic and nutrient source.

There are four significant municipal and industrial discharges within the study area. The location of these and other discharges are portrayed in Figure 20 and described in Appendix F. The locations of these discharges, estimates of their loads, and information regarding the times of their diversion to the spray-irrigation system are given in Table 10. The Whitehall Wastewater Treatment Plant and Plant Number One of the Misco Division of Howmett Corporation diverted their discharges on July 18, 1973; the Whitehall Leather Company diverted its discharge beginning November 24, 1974.



St. 201
Whitehall Waste Water Land Treatment Site
St. 227
St. 202
Old Treatment Plant
St. 203
St. 226
WHITE LAKE
St. 204
Lake Michigan
St. 205
Montague
White River
Coon Creek
St. 227
Certified Concrete, Inc.
Whitehall
Whitehall Leather Co.
Howmet Corp.
St. 203
Pin Key Mfg. Co.
Tech Cast Inc.
Hooker Chemical Co.
Pierson Drain
St. 226
Mill Creek
Wildcat Creek
St. 205
White Lake
St. 204
Birch Creek
Lake Michigan
Strawberry Creek
N
KEY
Industry
○ =in
● =out
Communities
//// = in system
\\\\ = out of system but to enter within 20 yr.
☐ = out, no plan
0 1 2
kilometers
△ station location

Figure 20. Location and diversion status of point sources and municipalities in the White Lake basin.

Table 10. ESTIMATED MUNICIPAL AND INDUSTRIAL LOADS TO WHITE LAKE PRIOR TO DIVERSION IN KILOGRAMS PER DAY*

Source & Location of Discharge	Receiving Water	Chemical Oxygen Demand	Biochemical Oxygen Demand	Suspended Solids	Soluble Reactive Phosphorus as P	Total Phosphorus as P	Dissolved Inorganic Nitrogen as N	Total Nitrogen as N	Date of Diversion	Source
Hooker Chemical Company, Montague	White Lake	949	--	1842	--	1.72	4.0	7.9		MWRC 1973
Misco Division of Howmett Corporation, Whitehall	White Lake	91.0	15.3	66.2	1.1	1.5	4.4	5.3	7/18/73	MWRC 1969 & 1970
Whitehall Leather Company, Whitehall	White Lake	1478	845	388	0	1.64	87.6	116	11/24/74	MWRC 1962, 1964, 1965, 1967
Whitehall Wastewater Treatment Plant, Whitehall	White River	--	13.5	128	13.8	14.1	29.0	32.2	7/18/73	MWRC 1967 & EPA 1975

*This table does not include minor business and industrial concerns in the area which have unmonitored small contributions. Also not included in this table are celery farm runoff, lagoon seepage, and septic tank drainage from Fruitland Township, Lakewood Club, Montague, Montague Township, and Whitehall Township. Other industries are listed in Appendix F.

Estimates of the nutrient loads to White Lake were obtained from operating reports and Michigan Water Resource Commission industrial and municipal waste surveys.

Tributary nutrient loads were calculated by multiplying concentrations of nutrients in the river by the specific daily flow as calculated from the USGS White River measurements (USGS 1971, 1972, 1973, 1974). Estimates of White River flows at the various stations were derived by multiplying the USGS measured flow by a factor to account for the increase in total drainage area. Estimates of the flows in Mill Pond Creek were obtained by multiplying the USGS White River gauged flow by a correlation factor derived from current measurements. The focus of the loading calculations was on phosphorus and nitrogen because they are believed to be the nutrients limiting algal growth in White Lake and are also the two parameters most likely to be affected by the diversion spray-irrigation project. A summary of the calculated loads from the White River and Mill Pond Creek is presented in Table 11. Plots of White River loads are given in Figures 21 through 24.

A comparison of nutrient loads from Mill Pond Creek with those of the White River demonstrate the insignificance of the smaller tributary. The White River load was over a thousand times greater than the Mill Pond Creek load. Since the water quality in Mill Pond Creek is similar to that of the other minor tributaries (MWRC, 1967), no further analysis was conducted on minor tributary loads. In subsequent discussions on lake loadings the minor tributaries were usually ignored.

Some patterns were observed in the seasonal variation of nutrient loads from the White River. Phosphorus loads were closely correlated with flow. During periods of high flow, concentrations increased due to higher non-point runoff, and the associated nutrient load showed a dramatic increase. Dissolved inorganic nitrogen loads were highest during winter periods when nitrate concentrations were high. Chloride and dissolved silicon loads showed a slight seasonal pattern with loads increasing with increases in flow.

One objective in calculating the various nutrient loads to White Lake was to separate the point and non-point sources, quantify their relative influences, and then to evaluate the potential and actual effectiveness of the wastewater diversion spray-irrigation system. Several additional computations were useful for this evaluation. Table 12 is an annual breakdown of the point versus non-point loads to White Lake. The point discharges were calculated from Table 10 and adjusted to reflect their period of actual operation. The non-point White River sources were more difficult to estimate because municipal discharges prior to diversion influenced the load determined at the river mouth. The non-point White River load was therefore estimated using loads calculated from Station 202. Corrections were made for differences in flow at the river station and at the mouth and for spray-irrigation drainage which began in August 1974

Table 11. ANNUAL AVERAGE WHITE LAKE TRIBUTARY LOADS IN KILOGRAMS PER DAY

Station	Year	Total Phosphorus as P	Nitrate + Ammonia as N
White River Station 227	1972*	45.5	194
	1973	59.9	223
	1974	88.4	314
	1975	80.1	424
White River Station 202	1972	31.4	165
	1973	55.8	185
	1974	70.5	248
	1975	71.9	376
White River Station 201	1972	30.6	392
	1973	52.1	211
	1974	80.7	267
	1975	75.7	386

*Estimated from known point sources and estimated non-point loads.

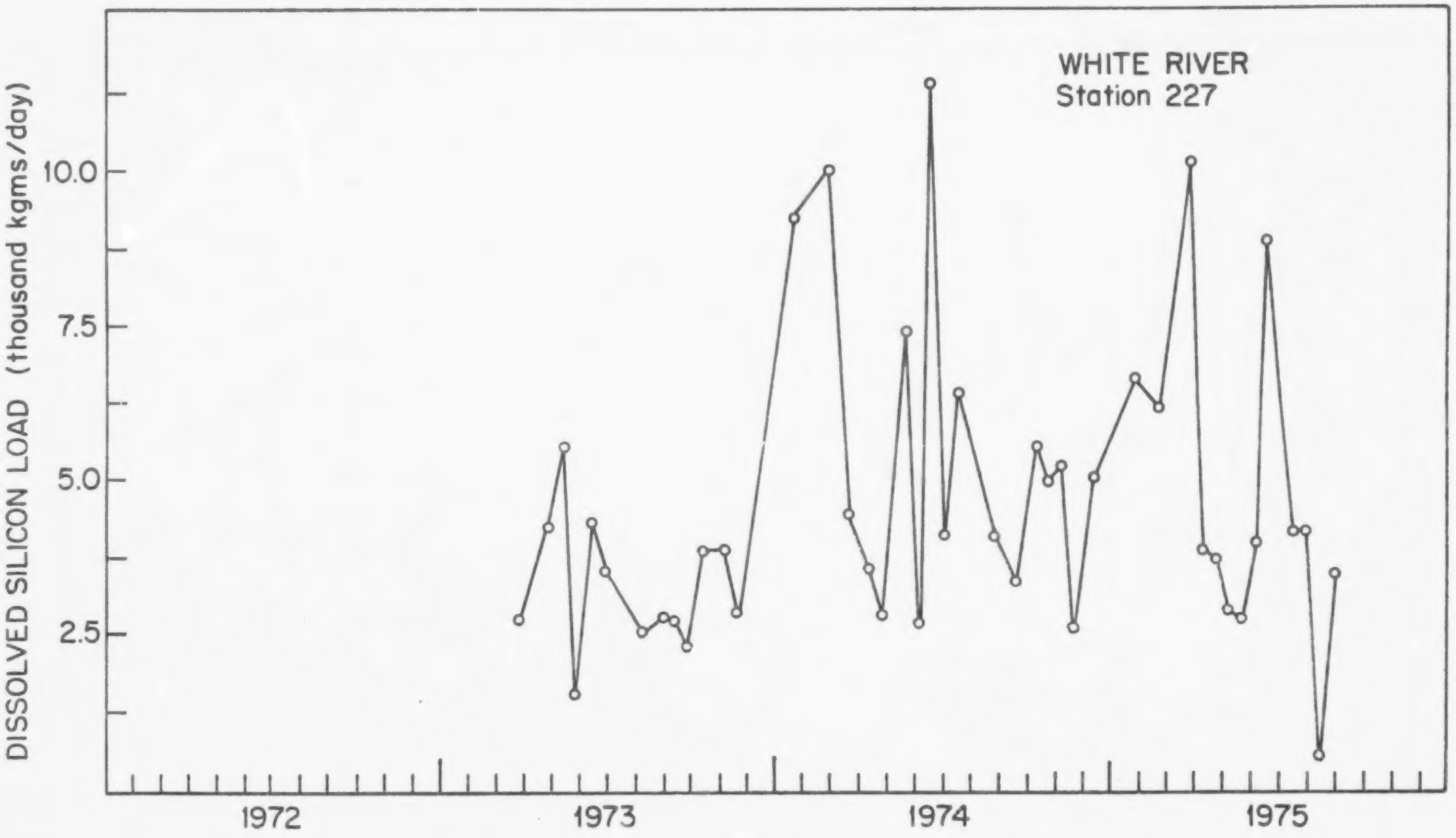


Figure 21. Silicon load of the White River, 1973-1975.

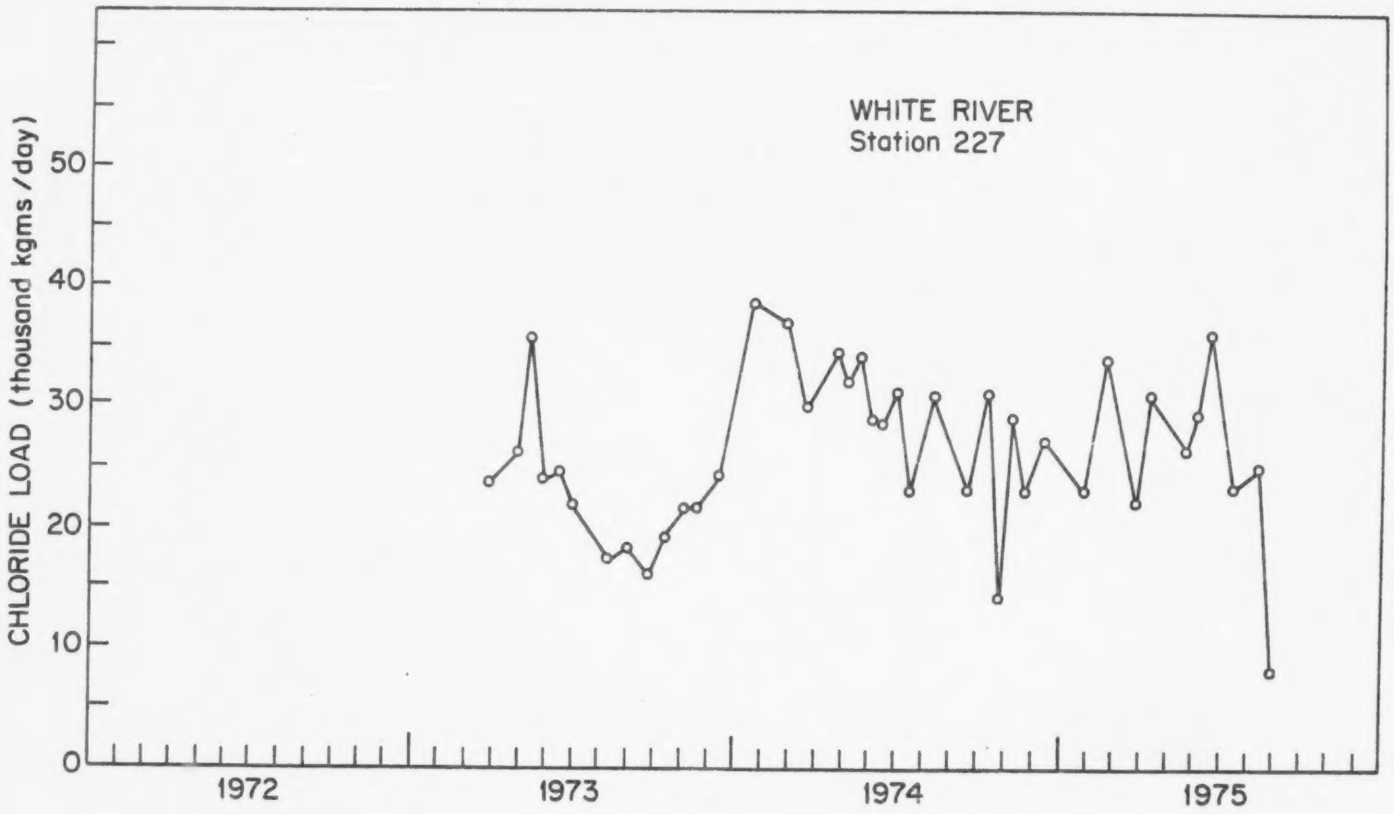


Figure 22. Chloride load of the White River, 1973-1975.

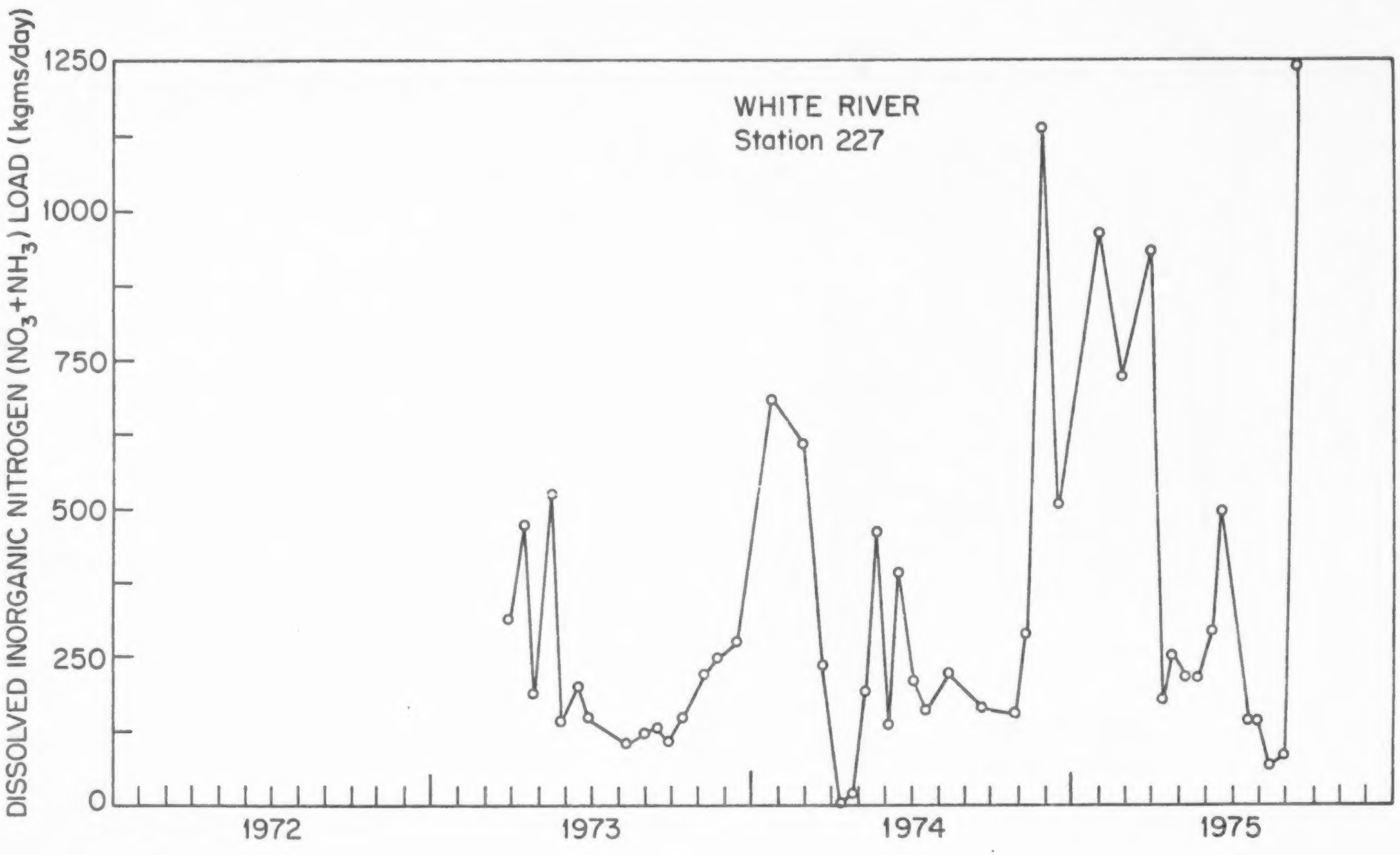


Figure 23. Dissolved inorganic nitrogen load of the White River, 1973-1975.

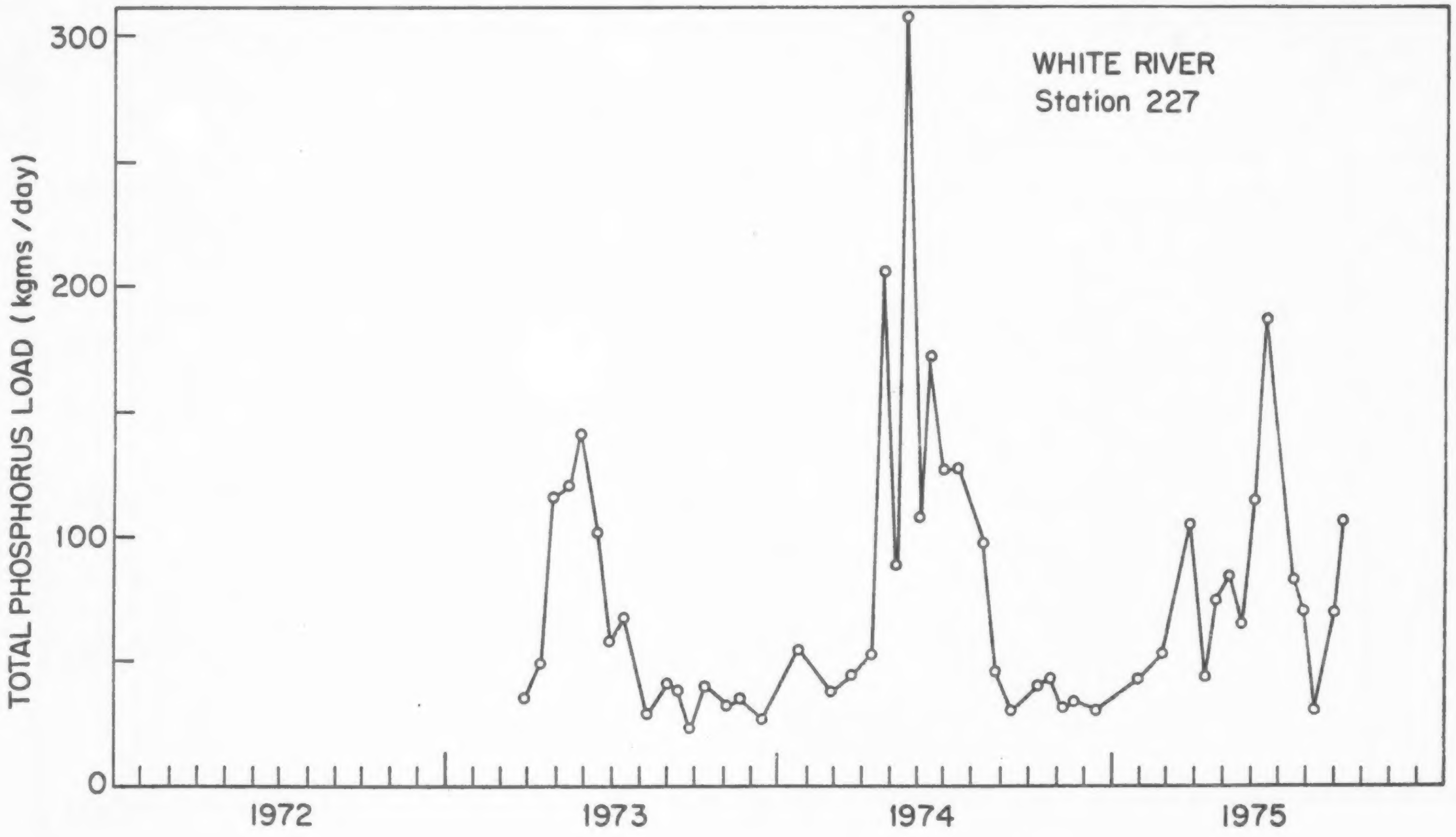


Figure 24. Total phosphorus load of the White River, 1973-1975.

Table 12. ESTIMATED TOTAL YEARLY NUTRIENT LOADS TO WHITE LAKE, 1972-1975 IN THOUSAND KILOGRAMS PER YEAR

	Total Phosphorus				Dissolved Inorganic Nitrogen			
Year	Municipal & Industrial	Non-Point White River	Spray-Site Drainage	Total	Municipal & Industrial	Non-Point White River	Spray-Site Drainage	Total
1972	6.92 (37.2%)	11.7 (62.8%)	0	18.6	45.6 (42.5%)	61.6 (57.5)	0	107.2
1973	4.13 (16.6%)	20.8 (83.4%)	0	24.9	40.1 (36.7%)	69.0 (73.3%)	0	109.1
1974	0.63 (1.9%)	32.2 (98.1%)	.002 (0)	32.8	30.7 (21.1%)	114.6 (89.9)	.002 (0)	145.3
1975	0.63 (2.1%)	29.05 (97.8%)	.015 (0.1%)	29.7	1.5 (0.9%)	154.67 (99%)	.033 (.1%)	156.2

(see Appendix E). The above calculations do not consider certain nutrient sources, such as septic tank drainage, minor tributary discharges, and atmospheric precipitation. Estimation of the contribution by these sources is difficult. Johnson (1975) measured atmospheric inputs of phosphorus to Saginaw Bay to be an average of 20 kg P/km^2/yr. Using this measured rate the atmospheric inputs of phosphorus to White Lake are calculated to be less than 1% of the total tributary load. A 1972-73 study by the USEPA on White Lake estimated the influence of minor sources to be less than 2% (USEPA, 1975a); these sources were therefore not included in further analysis.

A second calculation used in evaluating the changes in nutrient loads to White Lake was normalization of river loads to average flow conditions. Comparison of the loads between years can be confusing if there exist significant differences in total river flow because non-point sources are important. To normalize the loads for an average flow period, each yearly load was multiplied by the ratio of the 17-year average flow to the specific flow of each year. This method can normalize changes in flow but cannot normalize the effect of the changing non-point loads associated with the flow. The results of the normalized load calculations are given in Table 13.

The potential effectiveness of the sewage diversion can be estimated using the above calculations. If 100% of the 1972 point discharge loads to White Lake were eliminated, the expected reduction in the total phosphorus load would be 23% and for the dissolved inorganic nitrogen load it would be 31%. These estimates were obtained by averaging the non-point White River loads displayed in Table 12 and comparing these with the 1972 point discharges. If the normalized loading data are used, the respective percentage reductions are calculated to be 24.5% for total phosphorus and 27% for dissolved inorganic nitrogen. The nutrient load reduction expected to occur in the White River itself would be 19% for total phosphorus and 8% for dissolved inorganic nitrogen because all the loads shown in Table 12 do not discharge to the river (see Table 10). It should be noted that the reduction in phosphorus load is primarily with respect to the soluble reactive fraction. Almost 98% of the phosphorus discharged by the Whitehall Wastewater Treatment Plant was in this form. In contrast (see Table 9) only about 25-35% of the river phosphorus is in the soluble reactive form. The expected reduction in the White River soluble reactive phosphorus load would be approximately 50%.

The above comparison of municipal and industrial loads to the White River background loading is in good agreement with comparisons made from other data sources (MWRC, 1976, USEPA, 1975d). The Michigan WRC data indicate that total phosphorus point loads were approximately 20% of the total lake load and dissolved inorganic nitrogen point loads were 25% of the

Table 13. NORMALIZED YEARLY AVERAGE WHITE RIVER LOADS IN KILOGRAMS PER DAY

Station	Year	Total Phosphorus	Nitrate + Ammonia
White River Station 227	1972	51.7	226
	1973	55.7	207
	1974	61.0	217
	1975	60.8	322
White River Station 202	1972	36.7	193
	1973	51.9	172
	1974	48.6	171
	1975	59.2	286
White River Station 201	1972	35.8	458
	1973	48.4	196
	1974	55.7	184
	1975	57.5	293

total. Estimates derived from USEPA data suggest a similar reduction (20%) for total phosphorus. The USEPA data on total nitrogen indicated that point sources represented less than 15% of the total nitrogen load. Although the percentages between the various studies do not agree exactly (because of differences in sampling period and frequency and estimates of the point loads), the conclusions that can be drawn from each are similar. Even if the diversion is 100% effective, the reduction in nutrient loads to White Lake will not be dramatic. The exact percentage is difficult to determine because of variability in flow and non-point discharges.

To evaluate the actual effectiveness of the diversion spray-irrigation project a comparison among yearly average nutrient loads is necessary. An increase is seen in the total phosphorus and dissolved inorganic nitrogen loads when the raw load data for the river and the total lake examined (Tables 11 and 12). This increase is also seen, but to a lesser extent, in the normalized load data. However, the increase in total phosphorus and inorganic nitrogen loads observed is not a consequence of the spray-irrigation system. An examination of the upstream calculated loads at Stations 201 and 202 demonstrates that the observed increase is a reflection of increases in upstream loads. These loads do not originate within the study region which included the drainage from the spray-irrigation site. The spray-irrigation began in 1974; no observable effect on White River water quality was noted to result from the drainage. The observed increase in the upstream loadings for 1975 may have been because of peculiarities of the data available for that sampling year. The sampling program ended in October of 1975. The calculated 1975 loads were therefore biased towards periods of higher flow and nutrient loads. In addition, the sampling frequency in the winter and spring of 1975 was more thorough than in other years.

As was the case with inorganic nitrogen and total phosphorus, the yearly average silicon and chloride loads did not show decreases subsequent to implementation of the diversion project. Chloride yearly average loading fluctuated between 20 and 30 thousand kilograms per day. Dissolved silicon loads ranged between 3 and 5 thousand kilograms per day with some evidence of increased upstream loading subsequent to diversion.

In all prior discussions average yearly loadings were evaluated. This approach was used for several reasons including: 1) it provided the most statistically useful data set; 2) it was consistent with year-round lake sampling; 3) lake water quality problems (algal blooms) occurred during spring, summer, and fall periods; and 4) the effects of nutrient recycle and lake retention necessitated this approach. If loading data were evaluated seasonally, the calculated percent reduction from the sewage diversion project would be slightly different. The load data for total phosphorus for the months June, July, August, and September for 1972 to 1975 was averaged and compared to the diverted load. The average calculated reduction for this period was again approximately 25-30%. The percentage was much higher in 1972 and 1973 and lower in 1974 and 1975. This was a consequence of the varying upstream nutrient concentrations and flow. A similar calculation was made for the spring period. In this case the potential reduction in total phosphorus load to result from the diversion was approximately 20%.

In general, the measured nutrient loads to White Lake did not change as a result of the wastewater diversion spray-irrigation project. The anticipated reduction may have been masked by variability in upstream non-point loads. The baseline used in this study was 1972, a low flow year; other years of the study were high or moderate flow years. The flow normalization technique may not have been adequate in adjusting these loads to normalized conditions particularly with respect to total phosphorus. The anticipated dissolved inorganic nitrogen reduction of 8% in the White River was small in comparison to fluctuations in the overall load. Additional insight regarding the effectiveness of the diversion project can be gained by comparing calculations of the yearly, flow-weighted, average tributary concentration of nutrients, incorporating all of the point and non-point loads. The result from this calculation can give a rough indication of the potential effect of each yearly nutrient load on lake concentrations. It is not an accurate prediction of concentrations and ignores many phenomena such as mixing in the lake, physical, chemical and biological reactions, and various other source or sink mechanisms. The results from these calculations are presented in Table 14. These data further suggest that no large-scale improvement in water quality (nutrients, productivity, etc.) would be expected in White Lake during this investigation.

A more sophisticated analysis of the effect of nutrient loads on White Lake can be achieved using a model developed by Vollenweider (1975). This model was designed to relate nutrients (specifically total phosphorus loads) to trophic status. The details of the model are too complex for presentation in this report, however, the important conclusions of the model are depicted in Figure 25 and accompanying Table 15. The model correlates phosphorus loading rate in grams P/m^2/year to hydraulic load (the lake mean depth divided by the lake hydraulic retention time). The dangerous loading rate depicted on the graph represents the rate at which the receiving water would become or remain eutrophic. The permissible rate is a rate which would result in oligotrophic conditions. A load between the two is considered a mesotrophic rate.

The calculated average phosphorus loading rate for White Lake prior to diversion is approximately 2.4 grams P/m^2/year. At this loading rate the Vollenweider model suggests that eutrophic conditions would persist. Even the anticipated 25% reduction in total phosphorus load expected as a result of the diversion project (which has not yet been observed in the data) will not effect the trophic status of the lake according to

Table 14. FLOW WEIGHTED AVERAGE CONCENTRATIONS OF TOTAL PHOSPHORUS AND DISSOLVED INORGANIC NITROGEN IN TRIBUTARIES TO WHITE LAKE

Year	Total Phosphorus (μg P/ℓ)	Dissolved Inorganic Nitrogen (μg N/ℓ)
1972	45	259
1973	48	209
1974	47	210
1975	47	246

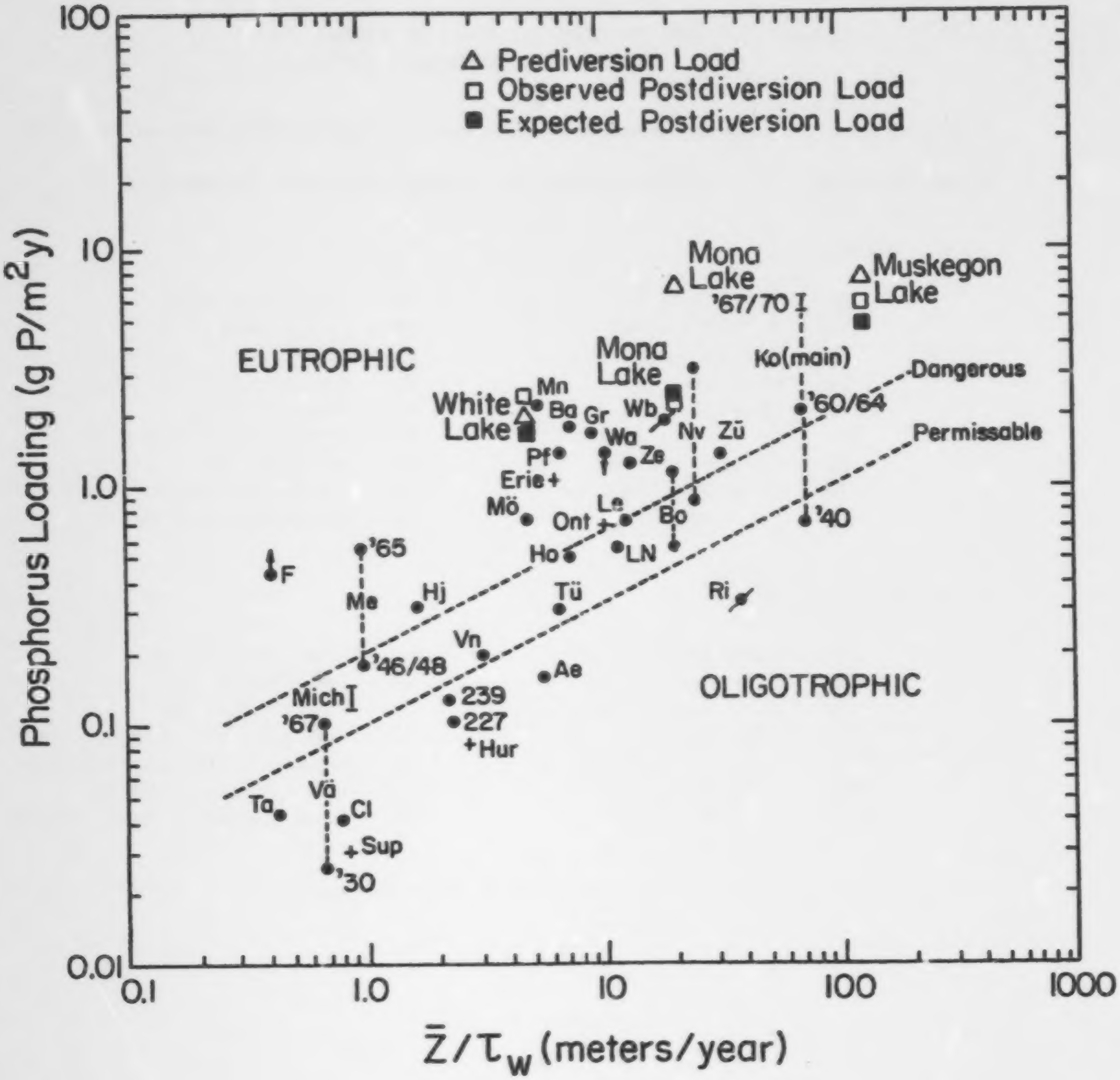


Figure 25. Vollenweider Model relating phosphorus loading per unit of surface to hydraulic load (mean depth divided by lake retention time) for Muskegon Lakes (Vollenweider, 1975).

Table 15. KEY TO SYMBOLS USED IN FIGURE 25 (AFTER VOLLENWEIDER, 1975)

Lakes Plotted: (o - oligotrophic; m - mestrophic; e - eutrophic)

1) Europe:

Ae - Aegerisee (o)
Tu - Turlersee (m)
Ha - Hallwilersee (e)
Bo - Bodensee (m)
Pf - Pfaffikersee (e)
Zu - Zurichsee (m-e)
Gr - Greifensee (e)
Ba - Baldeggersee (e)
Le - Lac Leman (m)

Ze - Zellersee (e)
LN - Lough Neagh (e)
F - Fures (e)
Va - Vattern (o)
Vn - Vanern (o-m)
Hj - Hjalmaren (e)
Ma - Malaren (e)
Wb - Wahnbachtalsperre (e)
Ri - Riverissperre (o)

2) North America:

Me - Lake Mendota (e)
Mn - Lake Monona (e)
Wa - Lake Washington (e)
Sup - Lake Superior (o)
Mich - Lake Michigan (o-m)
Hur - Lake Huron (o)
Erie - Lake Erie (e)

Ont - Lake Ontario (m)
Ta - Lake Tahoe (o)
Cl - Clear Lake (m)
227 - ELA Lake Nr. 227 (o; experimentally eutrophied)
239 - ELA Lake Nr. 239 (o)
Ko - Lake Kootenay (main basin, m-e)

the model. The model defined permissible loading rate is 0.73 grams P/m^2/year and the dangerous rate is defined as 1.43 grams P/m^2/year. Further, it can be noted from data displayed in Figure 25 that the phosphorus loading rate for White Lake is equal to or greater than rates of other lakes which have been classified as eutrophic.

In addition to calculations of nutrient loads to White Lake, estimates of the rate at which nutrients are leaving the system can be made by multiplying the measured White Lake channel concentration by the channel flow which is assumed equal to the total tributary flow. A summary of these flux calculations, including comparisons with nutrient loads to White Lake is presented in Table 16. In the case of total phosphorus a major percentage of this nutrient appears to remain in the lake although the results could be effected by intrusion from Lake Michigan. If this is true, then there must exist a significant pool of phosphorus at the bottom of White Lake which could have the potential for future nutrient release.

Summary -- The White River has been demonstrated to be the major hydrologic and nutrient source to White Lake. Estimates of the nutrient reduction expected to result from the diversion spray-irrigation project suggest a reduction in total phosphorus of 23% and dissolved inorganic nitrogen of approximately 31%. During the study period no reduction in nutrient loads or in tributary nutrient concentrations was measured. Reductions were not observed because of varying upstream non-point loads, the magnitude of which was equal to or greater than the point source reduction. The variation in upstream non-point loads is of sufficient magnitude to entirely mask the observed reduction. Future improvements in lake trophic status and water quality are only expected to be minor with respect to phosphorus and nitrogen. Reductions in loads for other water quality parameters such as heavy metals, organic matter, suspended solids, fecal coliform, and toxic organics were not investigated in this study.

No effect on water quality in the White River was observed as a result of spray-irrigation drainage. Nutrient concentrations flowing from the drain field were only slightly larger than those of the river. However, since this point source is relatively minor hydrologically, no effect on water quality was observed.

LAKE-RELATED CONSIDERATIONS

Spatial and Seasonal Distributions -- The presence of any horizontal variation in chemical characteristics in White Lake was examined by comparing total inorganic nitrogen and total dissolved phosphorus concentrations at a depth of 2 meters at Stations 203 and 204 (see Figures 26 and 27). With the exception of three points in the first year of study (1972), concentration patterns of total inorganic nitrogen and total

Table 16. YEARLY AVERAGE FLUX OF NUTRIENTS LEAVING AND ENTERING WHITE LAKE IN KILOGRAMS PER DAY

	Year	Total Phosphorus as P	Nitrate + Ammonia as N	Silicon as Si
White Lake Channel Station 205	1972	13.1	144	1276
	1973	16.1	264	2154
	1974	30.7	380	3371
	1975	19.9	401	2206
Total Tributary and Point Loads	1972	50.9	293	--
	1973	68.2	298	3473
	1974	89.8	397	5330
	1975	81.6	424	4705
Percent Retained in White Lake	1972	75%	51%	--
	1973	76%	12%	38%
	1974	66%	5%	37%
	1975	76%	5%	53%

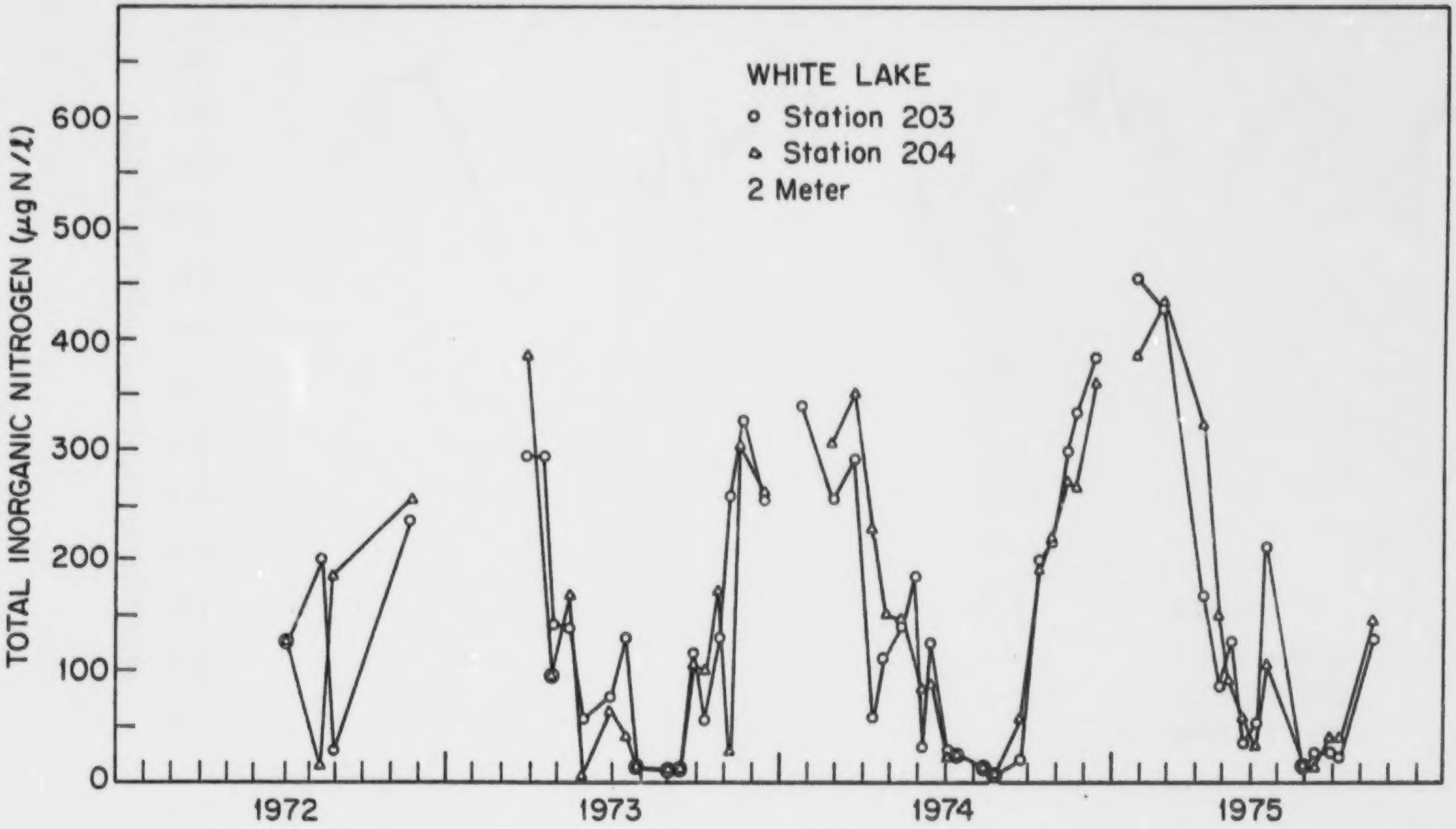


Figure 26. Dissolved inorganic nitrogen concentrations at Stations 203 and 204 in White Lake, 1972-1975.

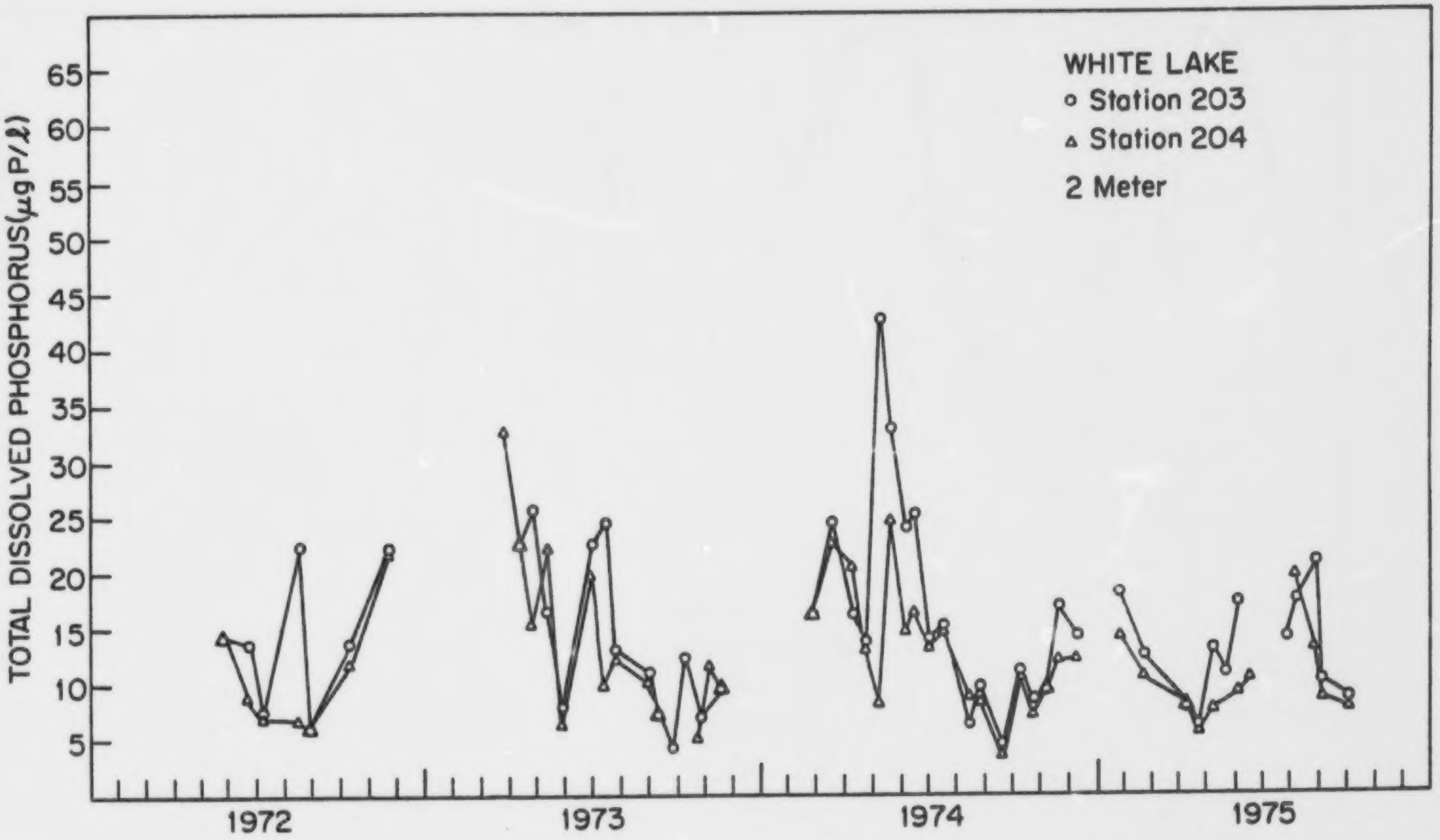


Figure 27. Total dissolved phosphorus concentrations at Stations 203 and 204 in White Lake, 1972-1975.

dissolved phosphorus followed each other very closely at both White Lake stations. Total dissolved phosphorus concentrations at Station 203 were usually greater than those at Station 204 although in most instances the difference amounted to less than 30%. The greatest spread occurred during May of 1974, when the total dissolved phosphorus concentration at Station 203 was almost two times that at Station 204. Less difference in total inorganic nitrogen concentrations was noted than in total dissolved phosphorus levels. Higher dissolved inorganic nutrient concentrations would be expected at Station 203, due to the influence of the White River. Later, as these materials move through the lake, they are transformed into organic nitrogen and phosphorus (via algal uptake), resulting in lower concentrations at Station 204. Due to the horizontal homogeneity of the system, no further discussion of horizontal distribution is considered necessary.

Vertical distributions in physical, chemical, and biological characteristics are closely tied to seasonal phenomena. The fluctuations in nitrogen, phosphorus, and silicon in the surface waters are largely a function of uptake by the phytoplankton. The phytoplankton populations, in turn, are governed by light, temperature, and the nutrient recharge associated with spring and fall thermal circulation periods. Concentrations of various chemical species in the bottom waters respond to changes in dissolved oxygen and oxidation-reduction potential; these, in turn, being influenced by the autochthonous supply of oxygen-demanding organic material provided by the plankton and macrophyte communities.

Surface water (2m) concentrations of nitrate ($NO_2^- + NO_3^-$) as shown in Figure 28 vary directly as a function of the phytoplankton populations. Maximum surface values of nitrate (200-400 μg N/ℓ) are recorded in the winter (December, January, February) and are rapidly depleted as a consequence of the spring (March, April, May) diatom blooms to near zero levels in the summer (July, August, September). After fall turnover, surface nitrate concentrations then increase towards their winter peak. Surface ammonia concentrations follow a pattern similar to that observed for surface nitrate (see Figure 29). Peak surface concentrations (200 μg N/ℓ) occur in December and January, due to a combination of high loads and low uptake, declining to a minimum in August and September. The reduction of total inorganic nitrogen to near-zero levels during the summer months (July, August, September) allows the nitrogen-fixing forms of the Cyanophyceae (blue-green algae) to dominate the phytoplankton community. Several of these algae are considered to be among the most noxious components of the plankton. The depletion of ammonia and nitrate during the spring and summer months indicates nitrogen limitation. Auer and Canale (1976) and USEPA (1975b), using bioassay techniques, found nitrogen to be limiting phytoplankton growth during this period. When nitrogen is depleted to zero in the surface waters of the lake, nitrogen-fixing forms of algae become dominant. Since these algae are able to fix atmospheric nitrogen, nitrogen should not, in a practical sense, be considered unavailable for growth during these periods. Phytoplankton growth is most probably limited by the availability of light or phosphorus.

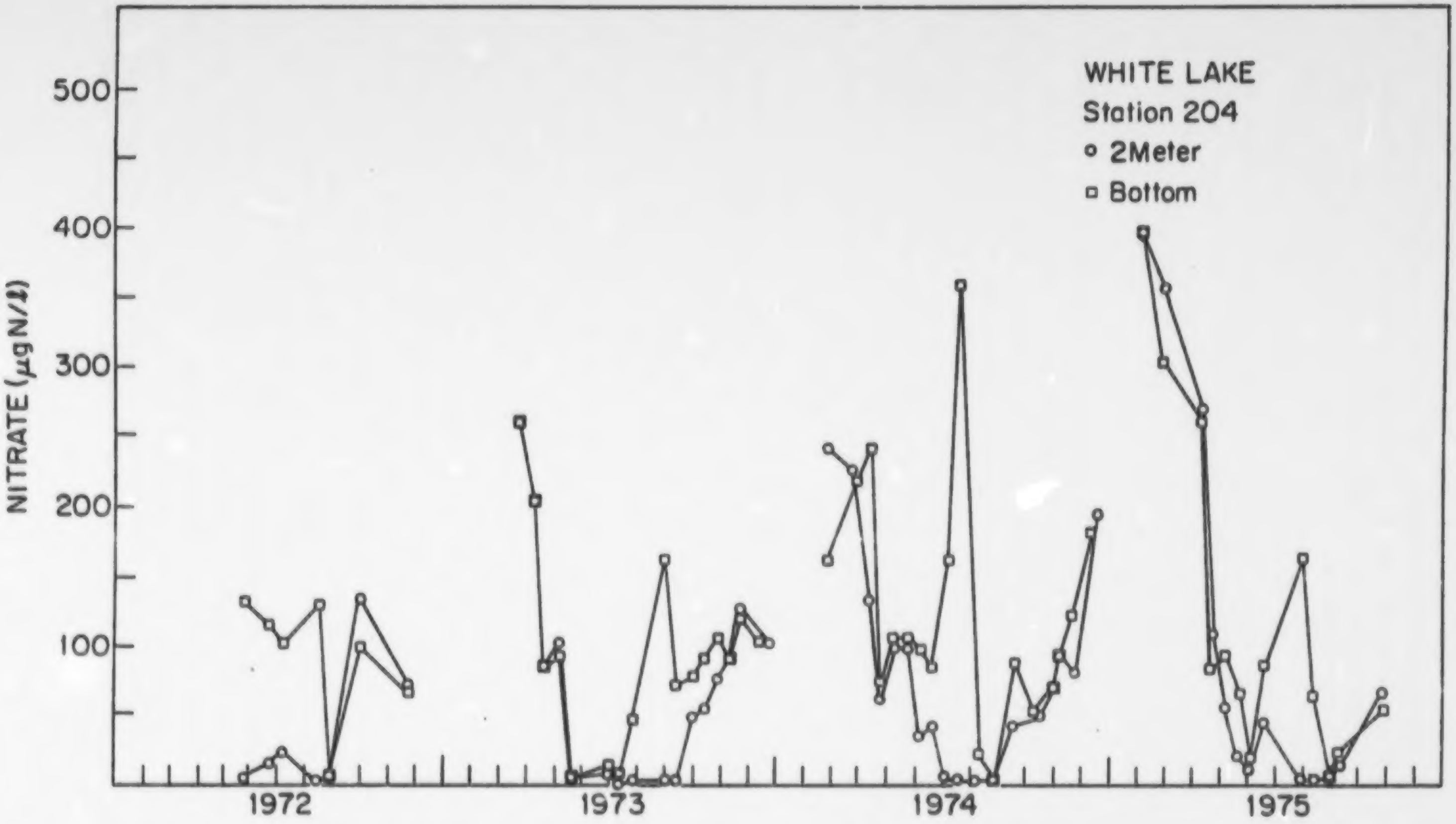


Figure 28. Nitrate concentrations in White Lake, 1972-1975.

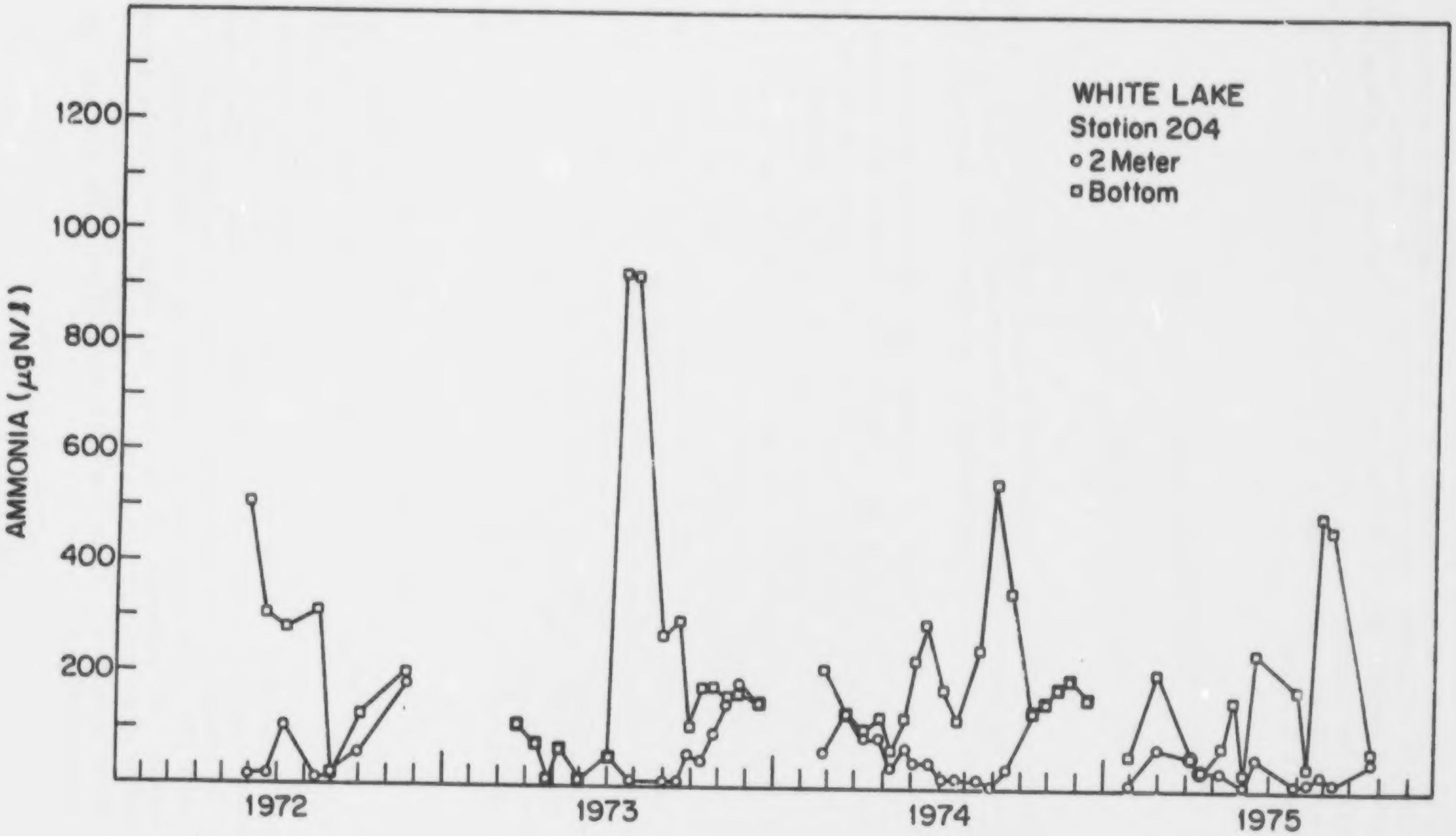


Figure 29. Ammonia concentrations in White Lake, 1972-1975.

At certain times of the year in the bottom waters of the lake, well below the photic zone, concentrations of dissolved inorganic nitrogen are dramatically different than those observed in the surface waters. Nitrate ($NO_2^- + NO_3^-$) levels in the bottom waters respond not to phytoplankton densities, but rather to the processes carried on by nitrifying bacteria. These organisms convert ammonia to nitrate with nitrite being an intermediate, as follows:

$$NH_3 \xrightarrow{\textit{Nitrosomonas} \text{ group}} NO_2^-$$

$$NO_2^- \xrightarrow{\textit{Nitrobacter} \text{ group}} NO_3^-$$

Since inorganic nitrogen production through ammonification and deamination processes exceeds uptake in the bottom waters, this region of the lake acts as a source of replenishment of dissolved inorganic nitrogen.

Bottom water concentrations of nitrate were observed to increase rapidly between December and March (see Figure 28). During this period concentrations of ammonia and dissolved oxygen are adequate to permit nitrification (see Figures 29 and 30). Bottom water nitrate concentrations are reduced in the period March-June as the bottom waters are mixed with nitrogen poor surface waters and the spring phytoplankton increase begins. Further nitrification in the bottom waters is halted at this point by the onset of hypolimnetic anoxia. It would be expected that nitrification would be non-existent in the bottom waters during the summer months, due to the lack of dissolved oxygen, regardless of high ammonia levels. Intrusion of well-oxygenated (and high-nitrate) waters from Lake Michigan, however, and the sporadic disruption of thermal stratification, provide sufficient oxygen to produce short-lived peaks of nitrate in the bottom waters during the summer months. With the occurrence of fall turnover, bottom water nitrate is mixed with the nitrate-poor waters of the epilimnion and assimilated by the phytoplankton, thus completing the seasonal cycle.

Evidence of intrusion was observed in the dissolved oxygen data for 1973 and 1974 and in the bottom water nitrate data for 1973-1975. This is depicted by the dashed line in the dissolved oxygen plot (see Figure 30). The dashed line represents the lowest concentration of dissolved oxygen in the hypolimnion whereas the solid line represents the concentration of dissolved oxygen 1 meter from the bottom. Higher dissolved oxygen at the bottom suggests intrusion of Lake Michigan waters. This explanation is also supported by the associated increase in bottom water nitrate (see Figure 28). The presence of nitrate in the bottom waters can also be related to the presence of hypolimnetic oxygen at depths greater than 1 meter from the bottom (which are designated in this report as bottom water samples). This prevents denitrification and will at times permit nitrification.

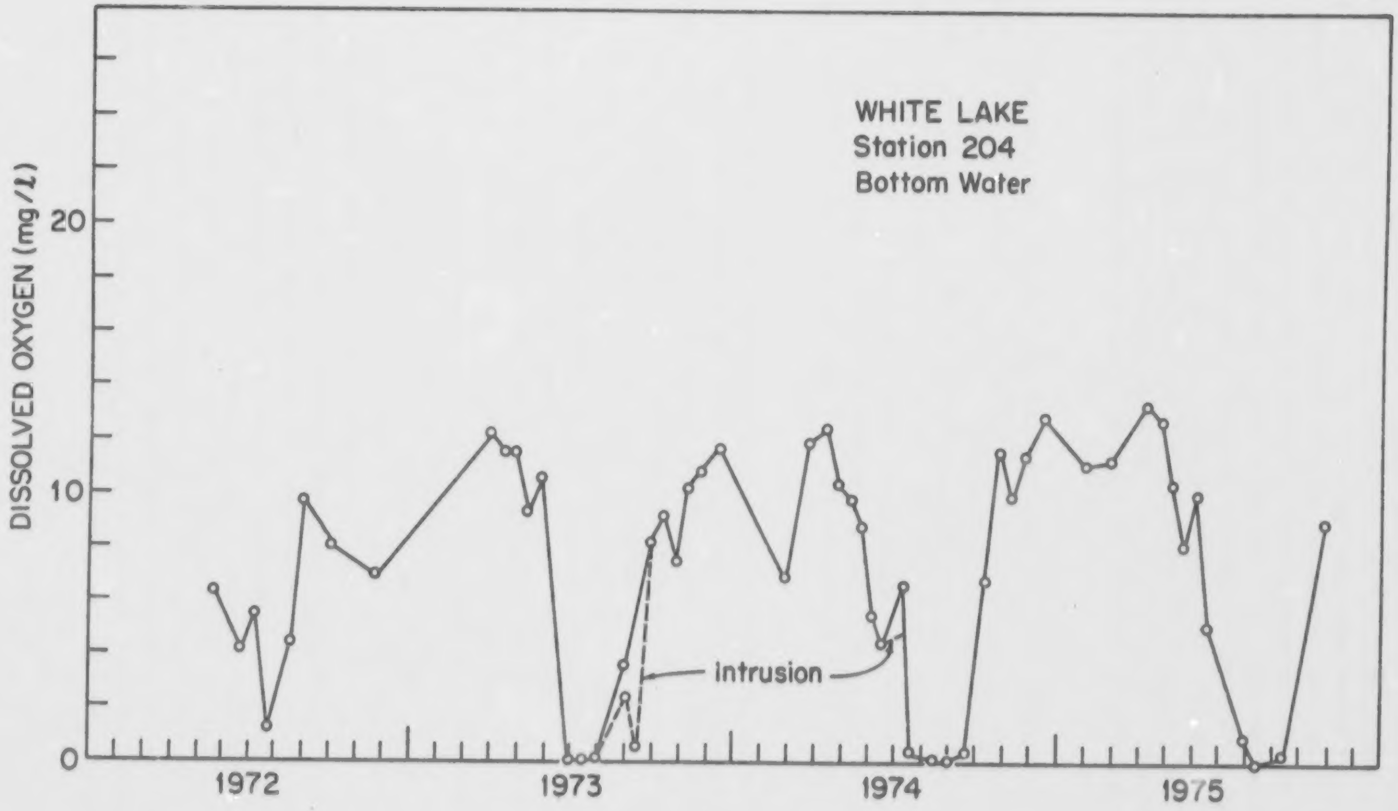


Figure 30. Dissolved oxygen in the bottom waters of White Lake, 1972-1975.

Ammonia concentrations in the bottom waters tend to provide a mirror image of the nitrate concentrations (see Figures 28 and 29). Highest levels of ammonia occur during the summer when decomposition processes are most active and nitrification is essentially halted by anoxia. The peaks in bottom water ammonia observed in October of 1974 and 1975 and in August of 1973 are the result of this process. Bottom water ammonia concentrations are severely reduced during the periods of nitrification (i.e., December-March). The nitrate cycle (depletion, decomposition, nitrification, and replenishment) in White Lake cause fluctuations of up to 250-400 μg N/ℓ in a given season. It was observed that ammonia concentrations in the lake were higher than those of the White River, the major tributary. This is due to the contribution to the nitrogen pool by nitrogen-fixing organisms in the lake and decomposition of organic nitrogen.

The phosphorus cycle in White Lake is, as was the case with nitrogen, tied to the seasonal cycles of dissolved oxygen and the phytoplankton. Additionally, dissolved inorganic phosphorus is closely related to the iron cycle. Three phosphorus fractions were measured in the Muskegon Lakes: soluble reactive phosphorus, total dissolved phosphorus, and total phosphorus. While many components of total dissolved phosphorus are ultimately utilizable by phytoplankton, only the soluble reactive phosphorus is believed to participate in precipitation reactions with iron.

Surface water soluble reactive phosphorus and total dissolved phosphorus (see Figures 31 and 32) follow similar seasonal patterns, although the latter is usually two to three times more abundant. This dominance is expected since soluble reactive phosphorus is more readily assimilated by phytoplankton. Variation in surface total dissolved and soluble reactive phosphorus is much less than that observed for the dissolved inorganic nitrogen species. Peak surface water levels of soluble reactive phosphorus and total dissolved phosphorus occur in late fall and winter, following the replenishment provided by turnover, the low winter productivity periods and spring loading. Surface levels of total dissolved and soluble reactive phosphorus are reduced rapidly from March to June, with soluble reactive phosphorus levels in the surface waters approaching zero in September due to uptake by the phytoplankton. Following this short period, increases in soluble reactive phosphorus were observed due to turnover. Auer and Canale (1976) and the USEPA (1975a) using bioassay techniques found phosphorus to be limiting phytoplankton growth in the fall. Concentrations of phosphorus then continue the cycle toward the winter maximum. Surface water total phosphorus levels reach a maximum during August and September (see Figure 33) suggesting both high levels of particulate phosphorus (organisms and detritus) and a possible transfer of the high concentrations of hypolimnetic phosphorus to the surface waters. Hypolimnetic peaks in total phosphorus are a consequence of sediment nutrient release. Summer peaks in surface water total phosphorus concentrations also appear to be largely a function of

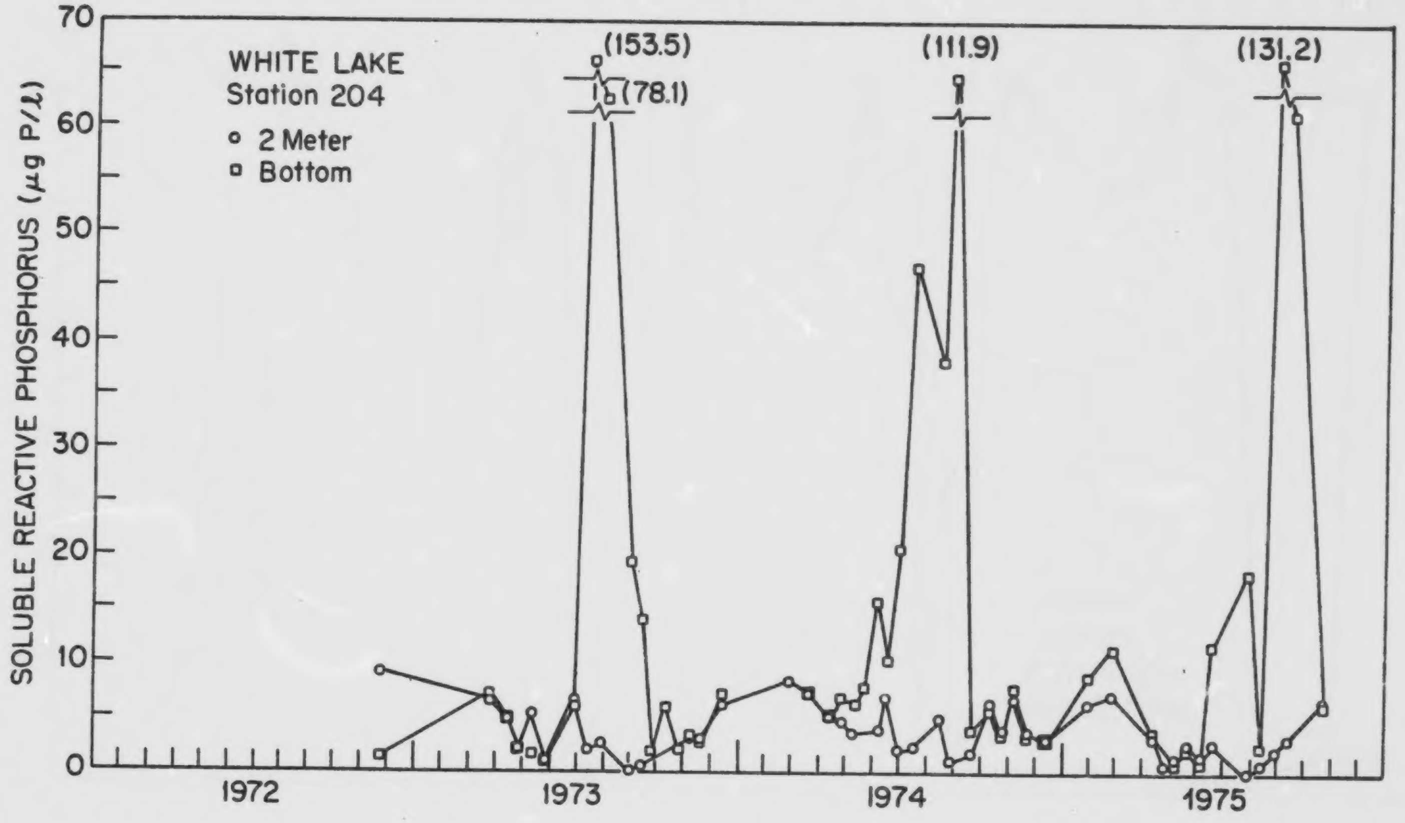


Figure 31. Soluble reactive phosphorus concentrations in White Lake, 1972-1975.

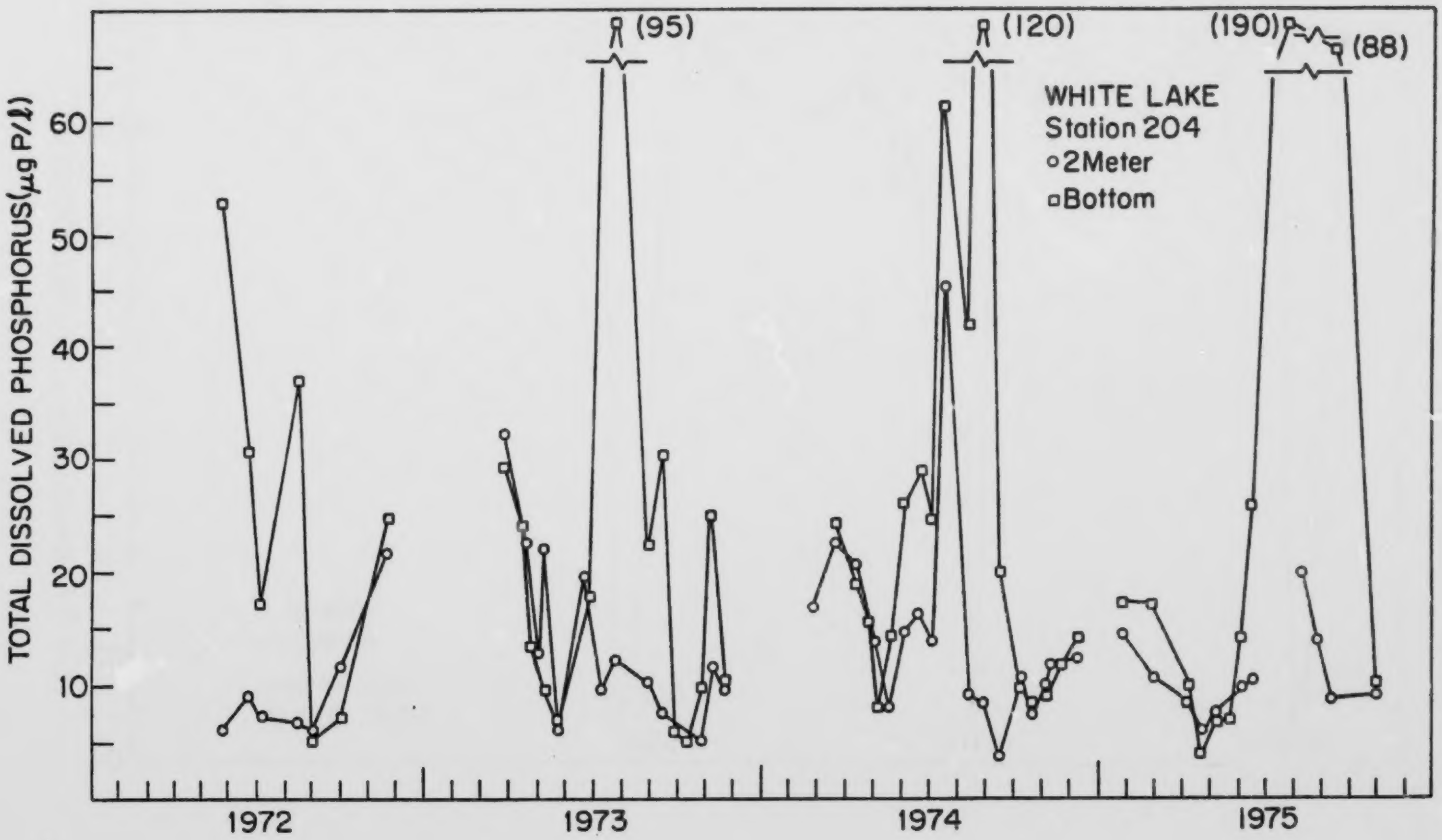


Figure 32. Total dissolved phosphorus concentrations in White Lake, 1972-1975.

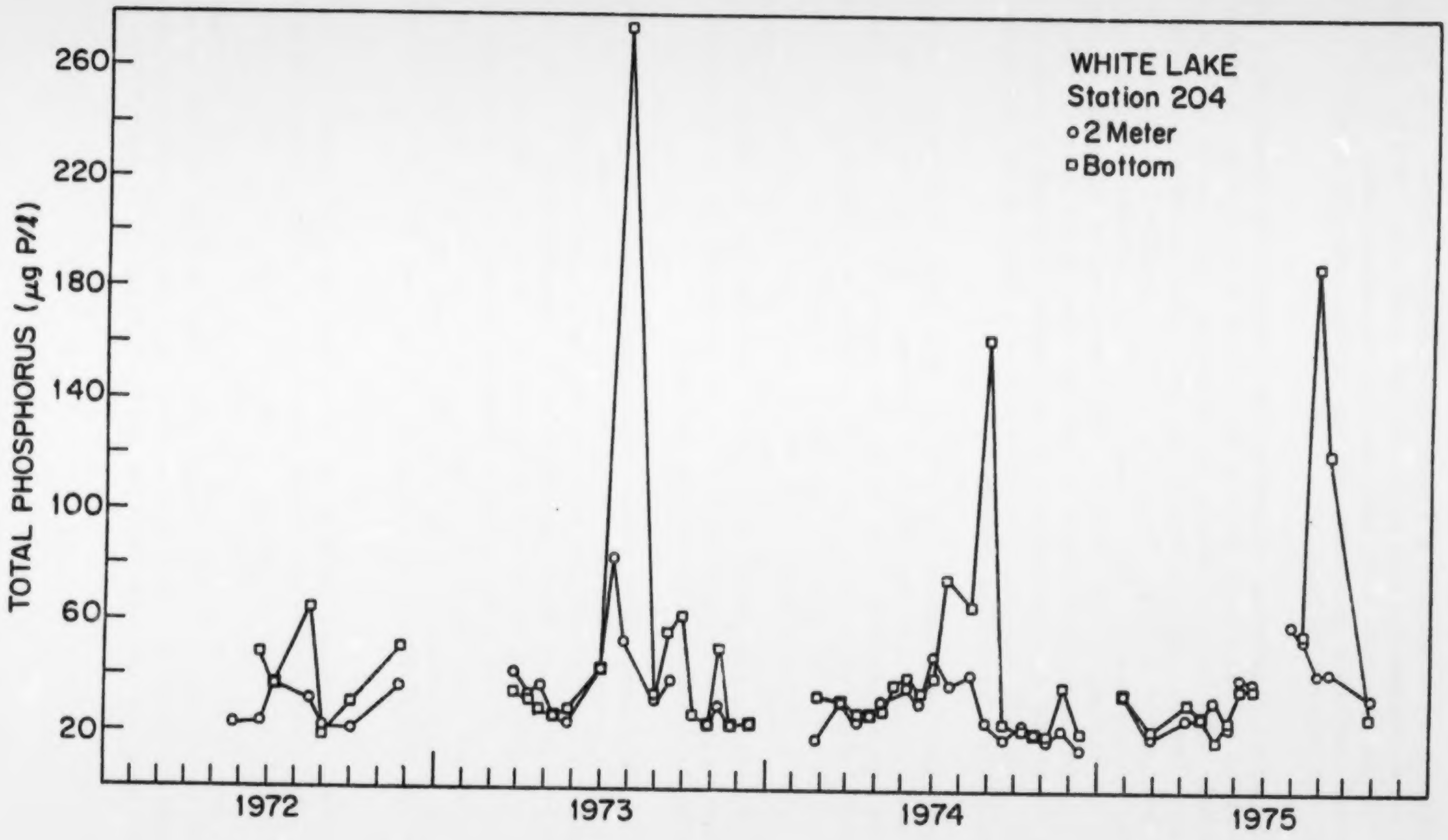


Figure 33. Total phosphorus concentrations in White Lake, 1972-1975.

of sediment release with a lesser dependence on loads. The White River peak total phosphorus concentration and load were observed in the spring and did not coincide with lake maximum concentrations. Further, the observed increases in summer concentrations were too rapid to be primarily attributed to loading.

Bottom water phosphorus concentration patterns are essentially identical to surface values except during the months July through September when extremely high concentrations are observed. The period when departure annually begins varies slightly but generally corresponds to the time of hypolimnetic anoxia. During stratified periods, phosphorus released to the bottom waters from decomposition and sediment exchange accumulates to extremely high levels. This increase is accentuated due to the limited exchange between strata and thus algal uptake. The fact that 70 to 90% of this phosphorus is present as soluble reactive phosphorus suggests the iron precipitation and dissolution phenomenon as a mechanism controlling transport.

Iron-phosphorus interactions are, at this time, imperfectly understood but may be generally explained as follows. During periods when the lake waters are oxygenated, iron is in the ferric (III) oxidation state. In this form iron is only sparingly soluble and reacts to form carbonate, hydroxide, or possibly phosphorus precipitates. If the compounds are formed, the phosphate ions may be adsorbed onto the surface of these materials. The result is the transfer of both iron and phosphorus from the soluble to the particulate form. Phosphorus released through the decomposition of organic matter in the sediment can be scavenged and trapped by oxidized iron in the sediment.

During the summer, when anoxic conditions prevail in the hypolimnion, iron is reduced to the more soluble ferrous (II) form. The compounds formed earlier are dissolved resulting in dramatic increases in the dissolved iron and dissolved phosphorus concentrations. Such occurrences were observed regularly during the summer in White Lake (see Figure 34). At the end of the summer, these chemical species are mixed through the water column (during fall turnover) and a portion of the soluble phosphorus is assimilated by the plankton. Some of the phosphorus is flushed from the system while the remainder reacts and is returned to the sediments.

Silicon is important because it is a major nutrient required for growth by certain phytoplankton, particularly the diatoms (Bacillariophyceae). These organisms are primarily of importance in White Lake during the spring and the fall. The silicon cycle in White Lake consists largely of a period of algal depletion in late winter and spring (February to June) with surface replenishment occurring in late fall and winter (November-February) (see Figure 35). Late summer surface concentrations of silicon are low and variable (0.5 - 1.0 mg Si/ℓ) indicating that

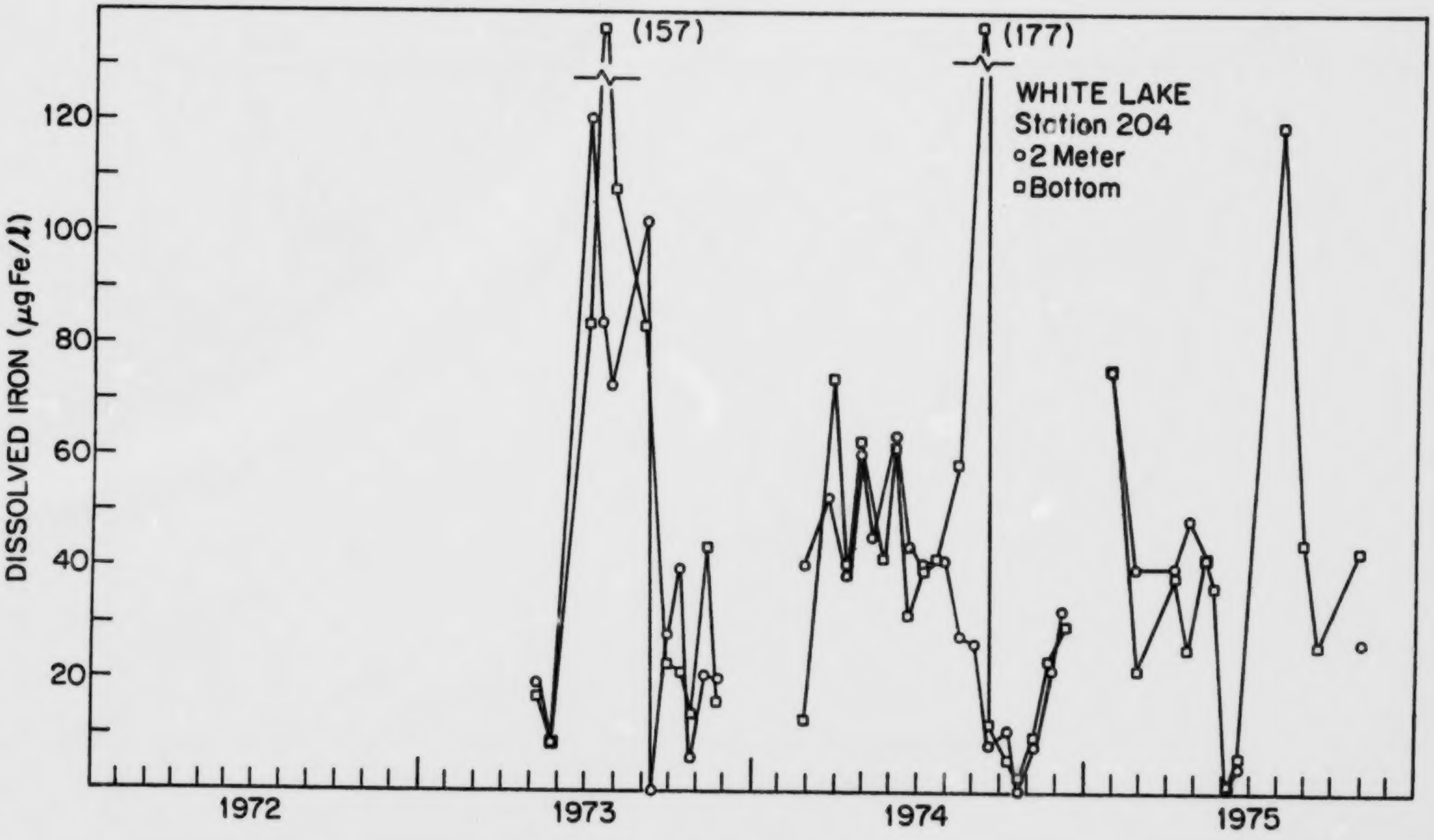


Figure 34. Dissolved iron concentrations in White Lake, 1972-1975.

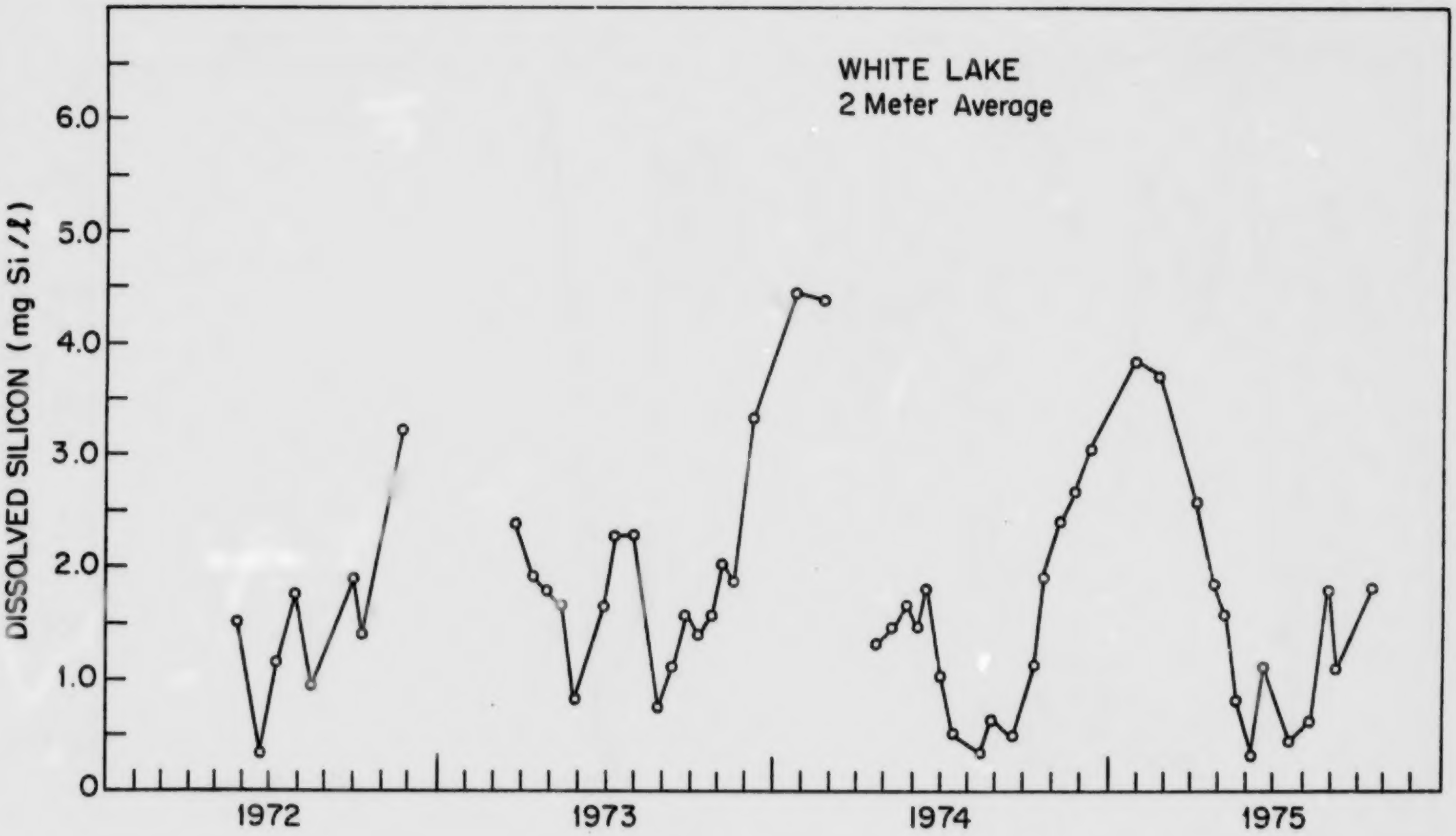


Figure 35. Dissolved silicon concentrations in White Lake, 1972-1975.

diatoms are an important component of the phytoplankton during this period. No severe depletion indicating a fall diatom peak was observed; the spring bloom on the other hand was significant. The failure of the phytoplankton populations to lower the levels of silicon closer to limiting values during the spring diatom bloom is unusual. This may be due to other types of growth limitation (e.g., nitrogen) or to the fact that rather high concentrations of silicon (4 - 4.5 mg Si/ℓ) are present during the winter months. Summer silicon concentrations are reduced below 0.5 mg/ℓ, but these levels are not believed to be growth limiting.

Chlorophyll *a* and primary productivity have been discussed in relation to nutrient cycles. The maximum values for chlorophyll *a* and primary productivity occurred simultaneously in the summer (see Figures 36 and 37). This is expected since the latter is in part a function of the former. The data indicate that peak phytoplankton populations for 1973 and 1974 occur during mid-July. The peak could not be determined with certainty for 1975 since there was no cruise during July; however, it appears that the highest chlorophyll levels may have occurred in September. The peak phytoplankton population is composed to a large extent to blue-green algae, a significant fraction of which are nitrogen-fixing forms. The abundance of nitrogen-fixing forms is related to the severe depletion of inorganic nitrogen in the surface waters. Michigan Department of Natural Resources (personal communication, 1976) reported that algicide was applied to White Lake in June of 1974. This action was not observed to have an effect on average chlorophyll levels.

An analysis of nutrient ratios can provide information regarding stoichiometric nutrient limitation of the phytoplankton in the lake. Stoichiometric nutrient limitation must be differentiated from actual growth limitation. If a nutrient is determined to be stoichiometrically limiting, this means that this will be the first nutrient which will become totally depleted if growth is unrestricted. However, it is possible for total depletion to fail to occur due to restraints imposed by time (seasonality or washout), light, temperature, algicides, etc. Only when the nutrient becomes depleted to a level below which phytoplankton uptake cannot occur will actual growth limiting conditions exist.
The average dissolved inorganic nitrogen to total dissolved phosphorus ratios (N/P ratio) can be calcualted for various circumstances. The winter N/P ratio in the surface waters varied from 17 to 32:1 between 1974 and 1975. For bottom waters the ratios were 37:1 (1974) and 25:1 (1975). Published estimates (Allen and Kramer, 1972) report that a typical non-limited ratio would be 10:1 to 13:1 by weight but occassionally as low as 3:1. Ratios larger than this indicate phosphorus limitation, lesser ratios indicate nitrogen limitation. By these calculations, White Lake would become phosphorus limited in the winter. The winter period was chosen as an example because of the characteristic low biological activity. In this circumstance, sources of available nutrients remain

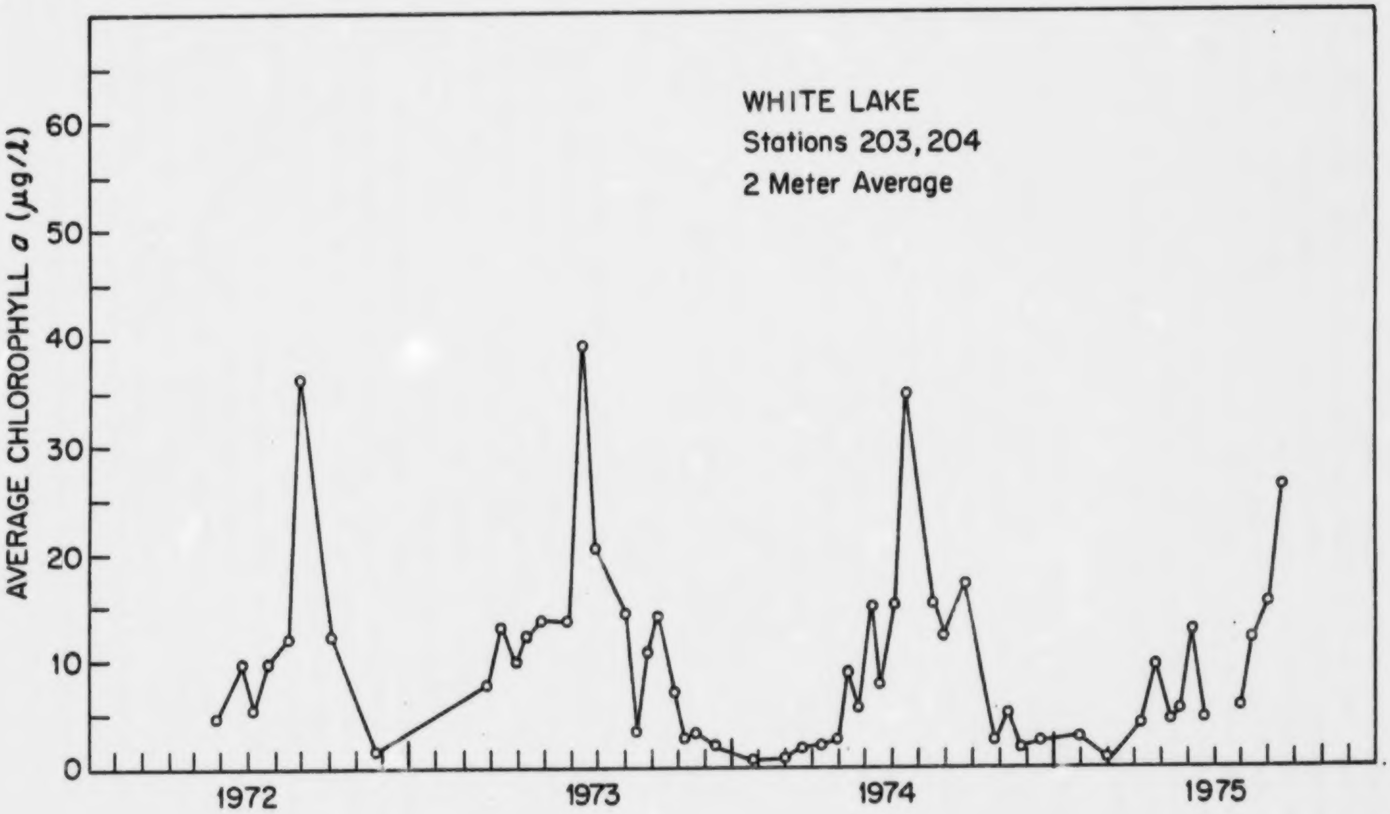


Figure 36. Chlorophyll *a* concentrations in White Lake, 1972-1975.

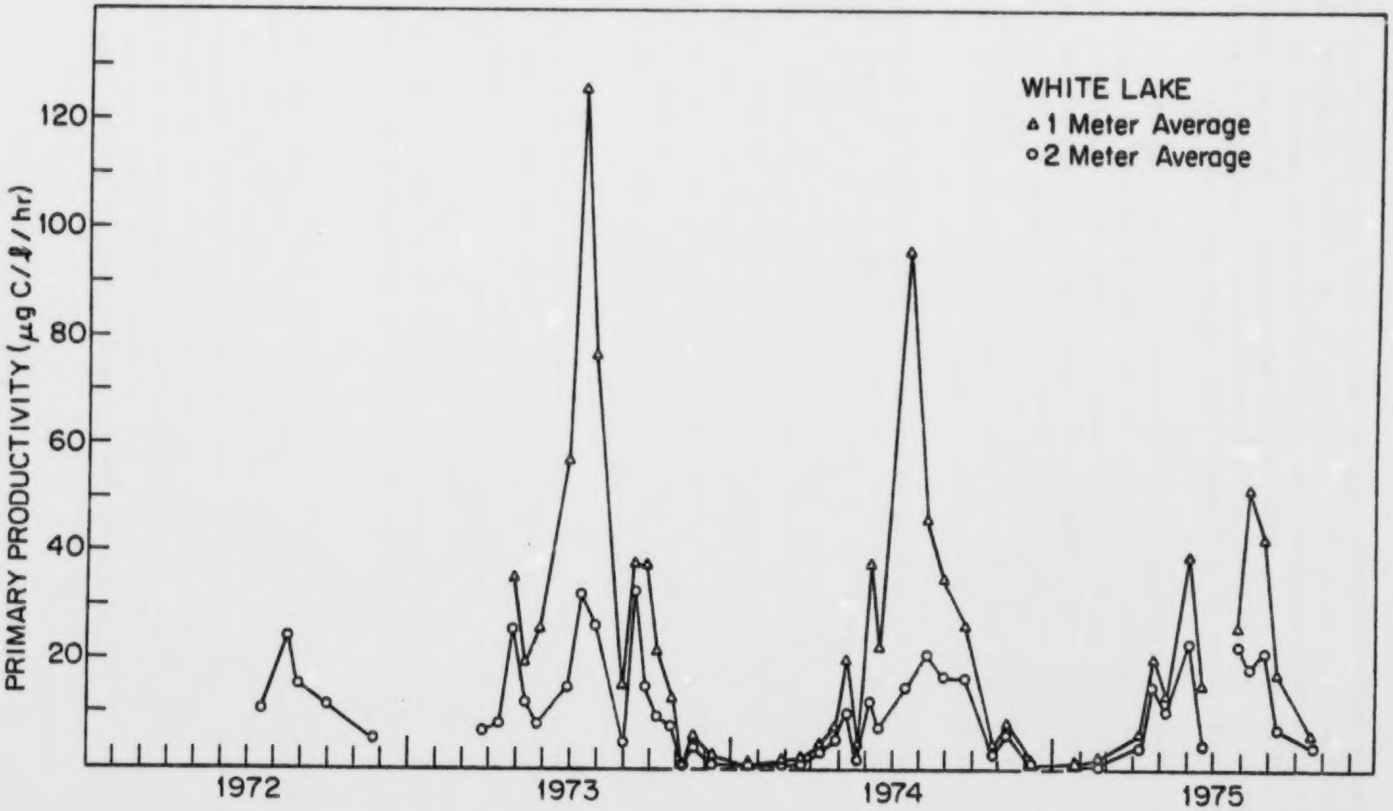


Figure 37. Primary productivity rates in White Lake, 1972-1975.

in their original form and analysis of the system is not complicated by uptake and conversion.

A more germane ratio would be that of summer surface N/P ratios. This ratio is difficult to calculate, however, since summer dissolved inorganic nitrogen levels approach the limit of detection because of algal uptake. In this case then, with dissolved inorganic nitrogen depleted, nitrogen would be found to be stoichiometrically limiting and growth limitation would occur if nitrogen-fixing algae were not present.

Another approach to nutrient ratio analysis is to develop the ratio of winter dissolved inorganic nitrogen in the bottom waters to summer total dissolved phosphorus in the bottom waters. This ratio describes the chemical conditions that might develop if there was no phytoplankton uptake and thus establishes a baseline nutrient status. Each of the two periods represents the maxima for the respective nutrient. For White Lake this ratio is 10:1 in 1974 and 7:1 in 1975. These ratios marginally indicate a relatively well-balanced nutrient supply.

A fourth approach to evaluating nutrient limitation through nutrient ratio analysis is to examine the ratios from the system loads. These average yearly calculations are 5.7:1, 6.28:1, and 8.6:1 for 1973, 1974, and 1975 respectively. These data suggest a well-balanced nutrient load or a possible tendency towards stoichiometric nitrogen limitation. The latter interpretation is supported by bioassay studies by Auer and Canale (1976) and the USEPA (1975b). A calculation of nutrient loading ratios for the period June through September (which excludes the periods of high dissolved inorganic nitrogen loading) show the ratios to be significantly less, on the order of 3:1. This further supports the hypothesis of summer stoichiometric nitrogen limitation.

The vertical distributions and seasonal variations observed for physical, chemical, and biological parameters in White Lake are consistent with what would be expected for a shallow, eutrophic lake in the temperate zone. One of the most impressive phenomenon observed for this, and the other Muskegon Lakes, is the tremendous seasonal fluctuations of nutrients in the surface and bottom waters. Variations in the magnitude of this flux should provide information regarding the effects of the diversion program on the lake.

Long-Term Changes -- This portion of the report deals with observations of long-term changes in physical, chemical, and biological parameters. It was anticipated by Chaiken *et al.* (1973) and Bauer Engineering (1971) that the diversion of wastewater from the Muskegon Lakes, followed by an on-land treatment and disposal system would result in an improvement in lake conditions (e.g., dissolved oxygen, nitrogen and phosphorus levels, phytoplankton densities). The objectives of this section are first to define trends in water quality and second to relate them, if possible, to the wastewater diversion.

The effects which could result from the diversion of wastewater would be the following: reduction of nitrogen and phosphorus concentrations resulting in lower phytoplankton populations and, ultimately, shorter periods of hypolimnetic anoxia. A reduction in the duration of hypolimnetic anoxia has benefits both with regard to direct effects on the biota and an influence on the flux of phosphorus from the sediments. To evaluate such changes, average annual values for various chemical and biological parameters at the surface and bottom waters were calculated (see Table 17). Annual averages were used as a principal tool for two basic reasons: 1) the annual average provided the most statistically valid data base and is not influenced by changes in yearly seasonal dynamics; and 2) water quality problems, e.g., algal blooms, occur in all ice-free seasons. Focus on one particular period would ignore others.

There were no observable consistent trends in the average surface or bottom water concentrations of soluble reactive, total dissolved, or total phosphorus in White Lake over the period 1972-1975 (see Table 17). Similarly, summer maxima of soluble reactive and total dissolved phosphorus in the bottom waters appear to be generally unchanged, although 1972 data are incomplete. The maximum summer values (both surface and bottom) for total phosphorus may have declined slightly over the three-year period. The surface and bottom water maxima are respectively for 1972, 49 and 127 μg P/ℓ; for 1973, 82 and 275 μg P/ℓ; for 1974, 75 and 136 μg P/ℓ; and for 1975, 60 and 190 μg P/ℓ. This change is a reflection of the particulate fraction since the dissolved fraction remains relatively unchanged. It can be said that any changes which may have occurred in phosphorus concentrations were, in relative magnitude, insufficient to dramatically affect the phytoplankton populations for the following reasons. The minimum summer surface total dissolved phosphorus concentration in 1975 was no lower than in previous years. Additionally, average total dissolved phosphorus concentrations in the surface and bottom waters remain unchanged from pre-diversion levels and are sufficient to support 10-15 μg/ℓ of chlorophyll *a* (assuming 1 μg Chl a/μg P).

More fluctuation was observed in average concentrations of dissolved inorganic nitrogen than for phosphorus. The most distinct trends were in bottom water concentrations of ammonia which decreased markedly from 373 μg N/ℓ in 1972 to 137 μg N/ℓ in 1975; nitrate concentrations in the bottom water varied, but with no discernable trend. Surface water concentrations of nitrate increased markedly over this period also from 56 μg N/ℓ in 1972 to 104 μg N/ℓ in 1975 while ammonia levels decreased from 69 μg N/ℓ to 37 μg N/ℓ during the same period. These changes may indicate an increase in oxidative conditions in the lake. Surface water levels of dissolved inorganic nitrogen continue to be depleted to near zero in the summer.

Table 17. AVERAGE ANNUAL VALUES FOR SELECTED WATER QUALITY PARAMETERS IN WHITE LAKE, 1972-1975

Year	Ammonia (µg N/ℓ)	Nitrate (µg N/ℓ)	Dissolved Inorganic Nitrogen (µg N/ℓ)	Total Dissolved Phosphorus (µg P/ℓ)	Total Phosphorus (µg P/ℓ)	Soluble Reactive Phosphorus (µg P/ℓ)	Chlorophyll *a* (µg/ℓ)	Secchi Disc (m)
				Surface				
1972	69	56	125	12	30	9	12.4	1.83
1973	69	76	145	14	39	5	11.4	1.92
1974	81	83	164	15	31	6	8.6	1.72
1975	37	104	141	13	40	4	8.6	1.79
				Bottom				
1972	373	110	483	34	51	6		
1973	195	92	287	31	53	21		
1974	192	116	308	25	39	15		
1975	137	84	221	30	52	17		

The occurrence of bottom water anoxia is an important aspect of the depressed water quality in White Lake. Although no significant or consistent reduction in the period of bottom water anoxia was observed in White Lake, an overall oxygen budget on the lake would likely reveal improved oxygen conditions. This is consistent with the ammonia and nitrate trends. The rate of bottom water deoxygenation shows a potentially decreasing trend during the study period. Thorough analyses of the oxygen resources of White Lake would require detailed studies on biochemical oxygen demand (BOD) and sediment oxygen demand. These were beyond the scope of this study. An improvement of oxygen conditions in White Lake might be a significant positive effect of the diversion of wastewater.

As mentioned earlier, the peak summer phytoplankton population could not be determined with certainty in 1975 due to sampling frequency. Consistent trends in phytoplankton density and activity were not observed in either the Secchi disc or primary productivity data. Although nutrient concentrations were relatively unaltered, improvements in average 2m chlorophyll *a* measurements were noted between 1972 and 1975 (12.4 to 8.6 μg/ℓ). There was no difference between 1974 and 1975. It is believed that nutrient concentrations have not been reduced sufficiently to be responsible for changes in phytoplankton densities.

Dillon and Rigler (1974, 1975) have developed relationships between spring total phosphorus at turnover and summer average chlorophyll *a* and chlorophyll *a* and Secchi disc transparency. Illustrations of these relationships are presented in Figures 38 and 39. According to these data, the average summer chlorophyll *a* levels in White Lake would need to be reduced by approximately 50 to 80% from the 1972 levels to gain a 1-meter increase in Secchi disc transparency. Furthermore, according to these calculations, an additional 30 to 50% reduction in total phosphorus levels would be necessary to achieve the required chlorophyll *a* reduction.

Summary -- The potential impact of the wastewater diversion on White Lake is largely a function of the percentage reduction in nitrogen and phosphorus loads which would be expected. The total phosphorus input to White Lake should decrease by 23% while the dissolved inorganic nitrogen reduction should be 31% as a result of the diversion project.

The actual average nutrient and water quality conditions for White Lake in 1972 and 1975 are presented in Figure 40 along with data from Lakes Michigan and Erie for comparison. Although nutrient concentrations in White Lake have not changed significantly, reductions in chlorophyll *a* were noted. Secchi disc depths remained unchanged. It would not be expected that the small alterations in the loads to the lake would have a dramatic effect on the trophic status, although the rate of hypolimnetic oxygen depletion may be decreasing.

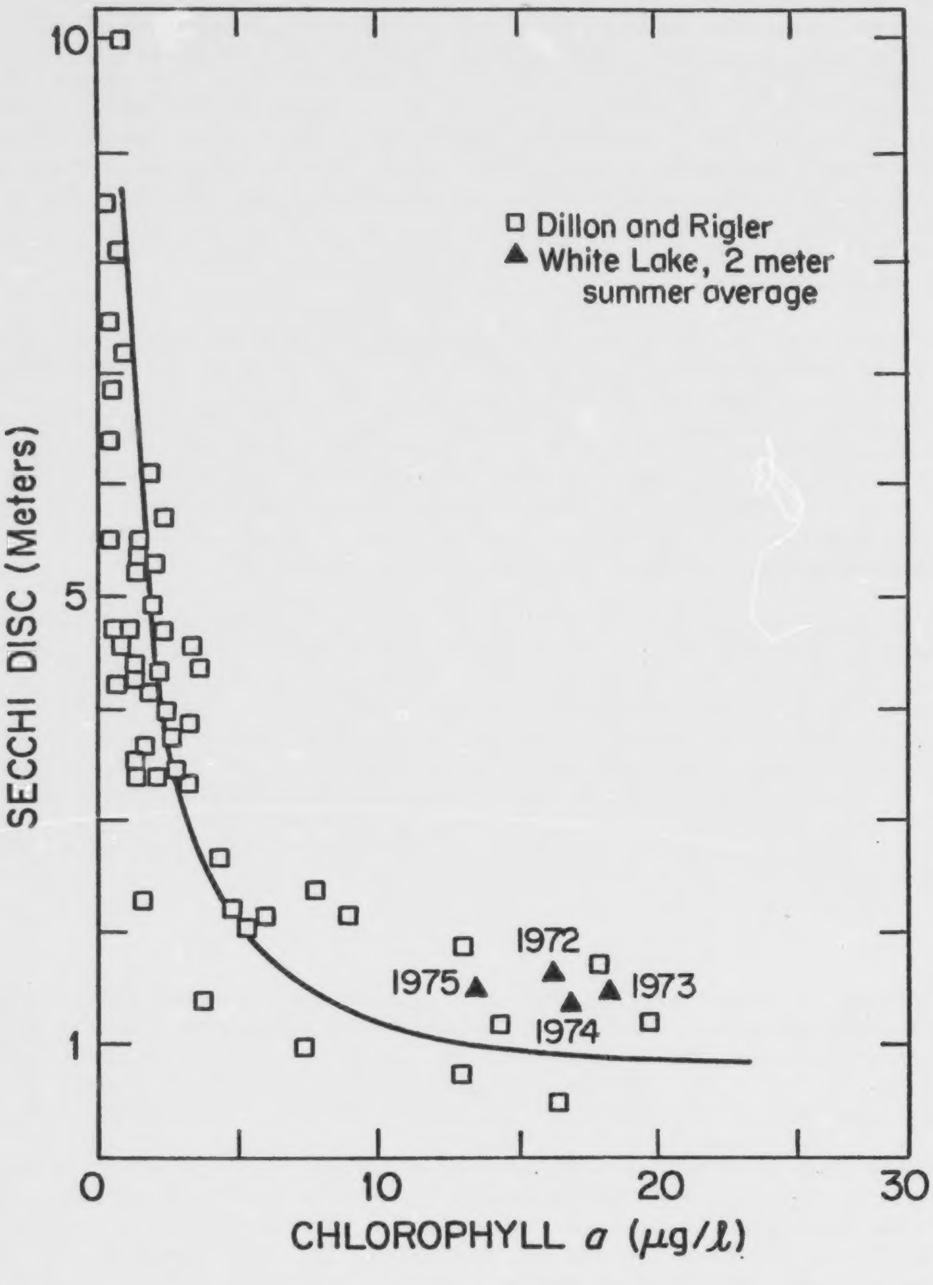


Figure 38. Correlation between chlorophyll *a* and Secchi disc (after Dillon and Rigler, 1975).

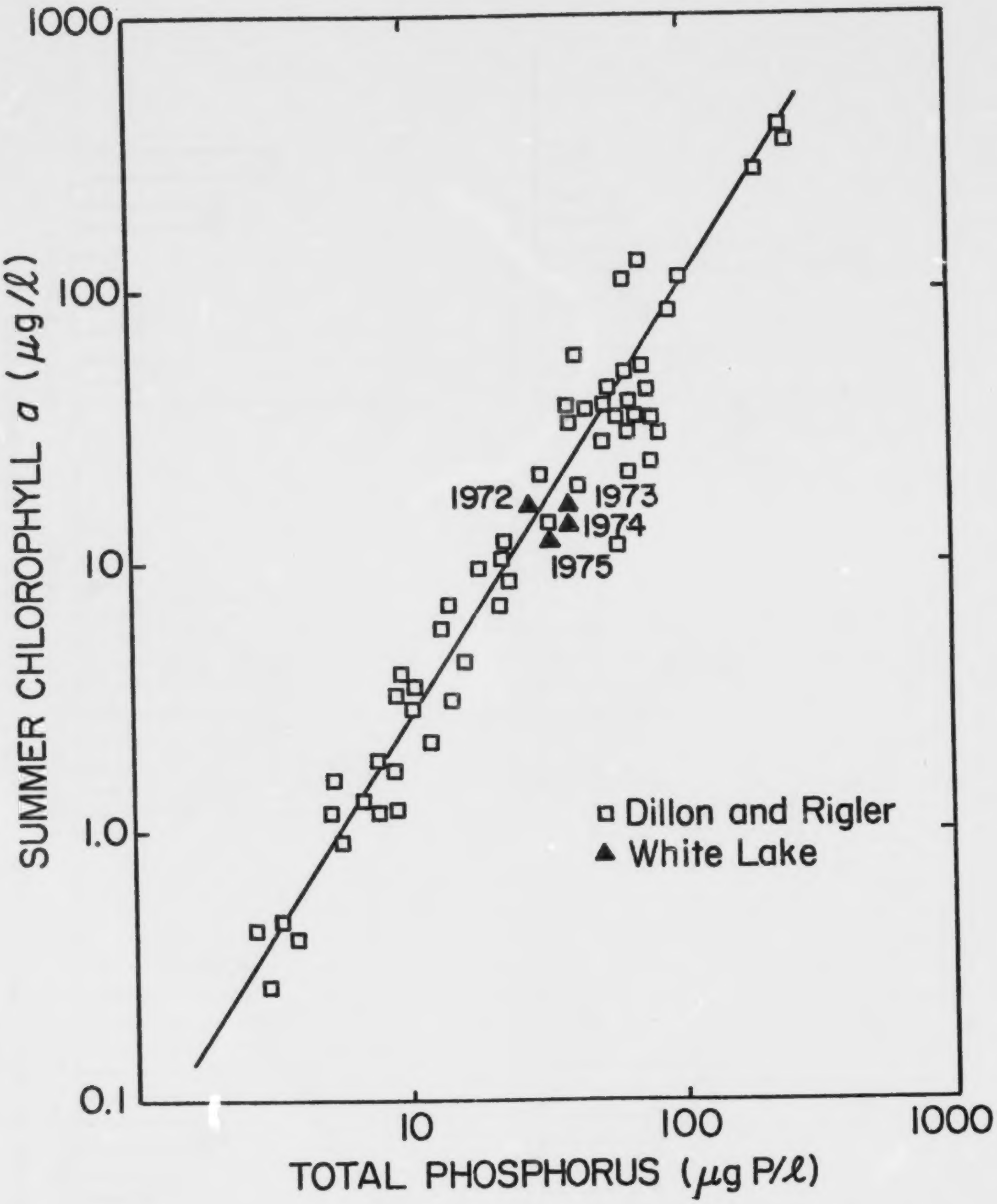


Figure 39. Correlation between phosphorus and summer chlorophyll *a* (after Dillon and Rigler, 1974).

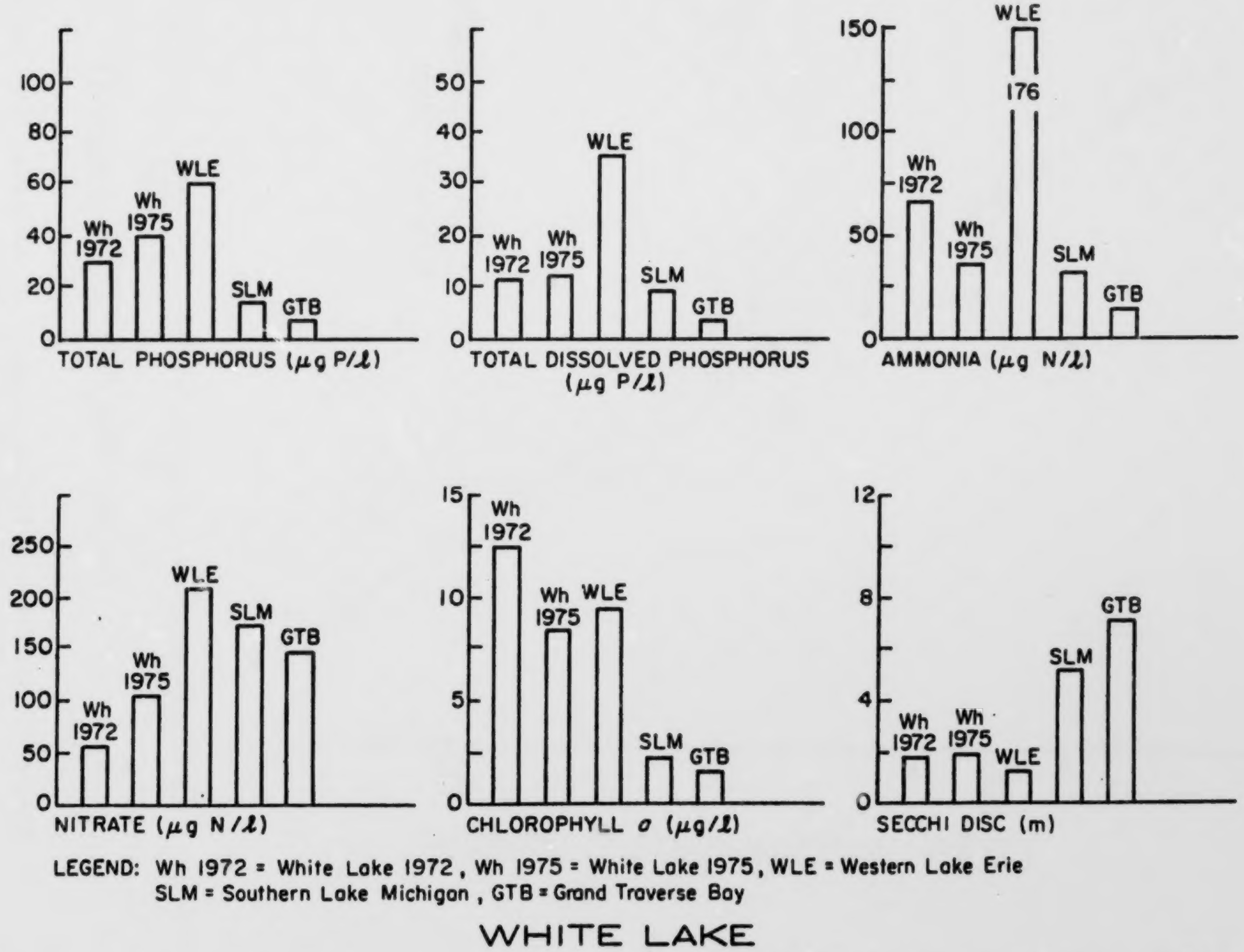


Figure 40. Pre- and post-diversion status of selected two-meter, yearly average water quality parameters in White Lake.

SPECIALTY STUDIES

Algal Growth Nutrient Bioassays -- Phytoplankton nutrient chemistry has been the subject of intense research efforts by a large segment of the scientific community. Several nutrients have been found to be of particular importance in limiting phytoplankton growth, among these: nitrogen (Thomas, 1970a, 1970b), phosphorus (Edmondson, 1972), carbon (Kerr *et al.*, 1970), iron (Schelske, 1962), silicon (Schelske and Stoermer, 1971), colbalt and molybdenum (Goldman, 1972), and manganese (Shapiro and Glass, 1975). It seems logical then to suggest that if the chemistry of certain of these nutrients is understood, and the state of nutrient limitation in a particular lake is known, manipulation of nutrient concentrations should result in control of the phytoplankton population. This is, of course, the case as has been demonstrated in a number of instances, the most notable being that of Lake Washington (Edmondson, 1972).

Preliminary to the manipulation of nutrient concentrations, a study of nutrient limitation in the lake phytoplankton population is necessary. The algal growth nutrient bioassay provides a method for such a determination. Algal bioassays are conducted for the purpose of determining the nutrient status of algal populations. Generally, it is wished to determine which nutrient, if any, limits the growth of that population. In most bioassays, a series of nutrient solutions are added to separate aliquots of lake water and the resultant growth is measured. The element, whose addition stimulates growth to the greatest extent, is described as the limiting nutrient. The following is a summary of nutrient bioassay studies conducted on White Lake. A full discussion of these results is given in Appendix A.

Water samples were collected from White Lake (Station 203) at intervals chosen to represent seasonal variations (November 2, 1973; April 27, 1974; July 18, 1974; and October 25, 1974). The samples were filtered to reduce the numbers of predatory organisms (zooplankton). The lake water, with its natural assemblage of phytoplankton, was placed in each of six 14-liter pyrex carboys. Each carboy, with the exception of the control, received a nutrient addition: ammonia, nitrate, ammonia plus nitrate, phosphorus, or a trace metal mixture. The carboys were outfitted with submerged air diffusers and magnetic stirrers. Each reactor was placed in a constant temperature bath equipped with a fluorescent light bank. The lighting was provided in a diurnal cycle. Samples were taken daily for chlorophyll *a*, pH, temperature, and chemistry. Samples of water in the reaction vessels were taken, typically every third day, and analyzed for cell enumeration and identification. A more detailed discussion of the experimental methods is contained in Appendix A.

The addition of orthophosphate phosphorus to White Lake water stimulated algal growth markedly, particularly during the fall (see Figure 41). The addition of nitrate nitrogen to White Lake water resulted in increased algal growth (*Fragilaria, Cyclotella, Asterionella*) in the spring which had a longer and more evenly distributed nature than did additions of phosphorus (see Figure 42). While the magnitude of growth was not as high with nitrogen as with phosphorus additions, neither were the crashes as severe. Trace metals appeared to be associated with the enhancement

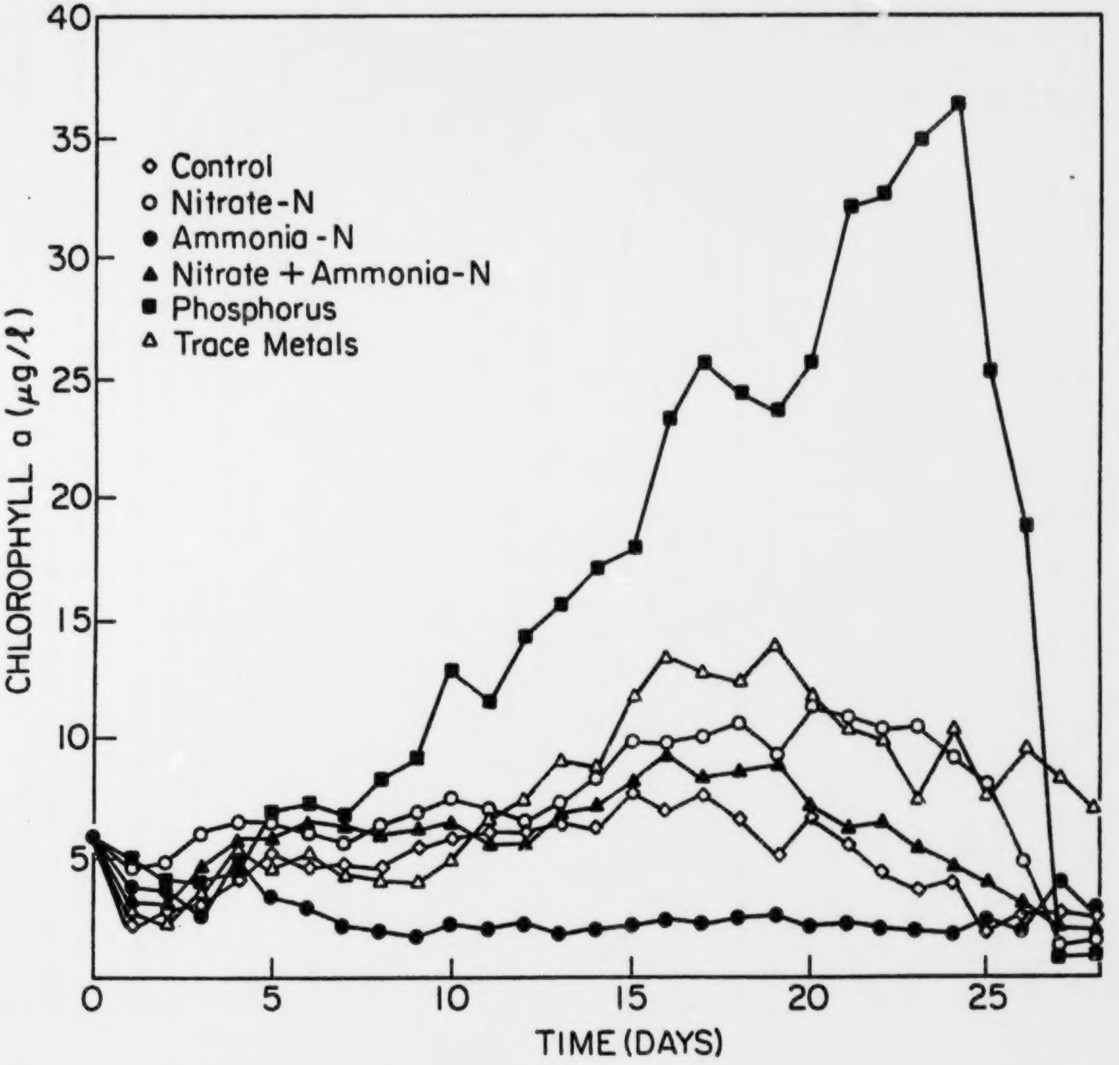


Figure 41. Chlorophyll *a* concentrations - 2 November 1973 bioassay.

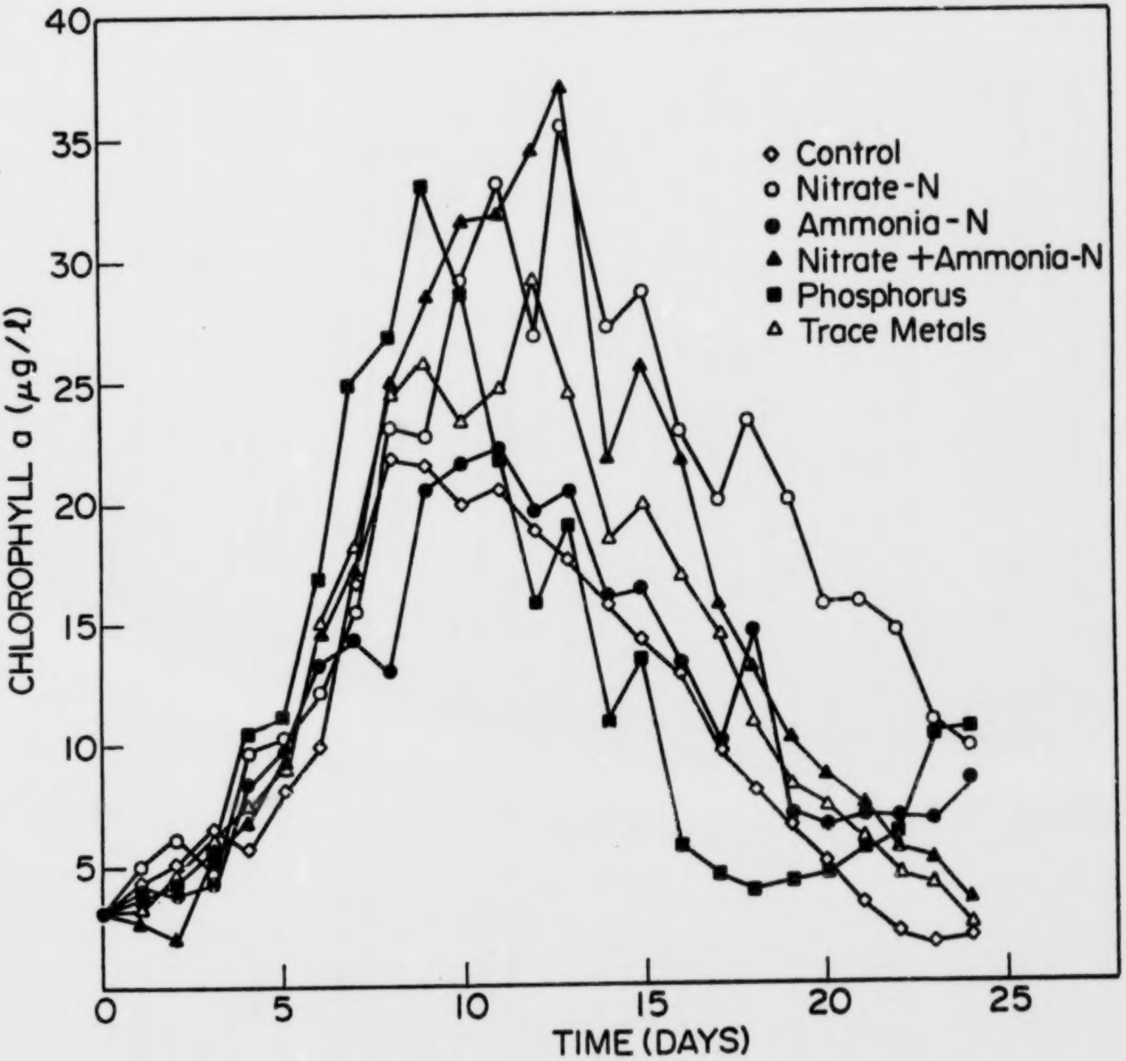


Figure 42. Chlorophyll *a* concentrations - 27 April 1974 bioassay.

of the metabolism of available nitrogen. The addition of trace metals mixture also increased the algal uptake of nitrogen over the period of the bioassay. The addition of phosphorus usually resulted in rapid population peaks of *Fragilaria* and *Melosira*. The species of these genera observed to bloom are considered opportunists and would not be expected to dominate following nutrient reduction. Nitrogen limitation was most important during the spring and summer. The dominant phytoplankton in the summer in White Lake are nitrogen-fixing forms and therefore, reductions in phosphorus will be required before these species can be controlled.

Ammonia nitrogen was, in all cases, found to be inferior to nitrate nitrogen as a nitrogen source. In reactors receiving ammonia and exhibiting nitrification, growth was often inhibited even with copious quanities of nitrification-produced nitrate nitrogen available. The nitrogen uptake and transformation associated with the nitrifying organisms was very much greater than that associated with algal populations.

Macrophytes -- Extensive research has been directed towards evaluating various aspects of aquatic macrophyte ecology in lakes and other systems (Westlake, 1968). This research, however, has not adequately described the significance of macrophytes to total lake ecology in a broadly applicable manner. The significance of macrophytes to lake nutrient budgets and cycling remains unclear. The nutrient content of plant tissues has been analyzed by several investigators. Although wide variability in nitrogen and phosphorus concentrations have been reported (Caines, 1965, Anderson *et al.*, 1966, Gerloff, 1973, Neel *et al.*, 1973) aquatic plants generally average about 3% nitrogen and 0.4% phosphorus as dry weight. Results from laboratory studies by DeMarte and Hartman (1974) suggest that aquatic plants can play a significant role in recycling nutrients within aquatic systems. This occurs not only when the plants die and decompose but also during periods of active growth. Such an occurrence is the result of the active transport of nutrients along the substrate-root-stem-leaf-water pathway. Data obtained by Neel *et al.* (1973) from plant tissue meausrements suggest, however, that in Lake Sallie, Minnesota, macrophyte control programs such as harvesting, would not be effective in decreasing nutrient concentrations in the lake. Macrophytic nitrogen and phosphorus were insignificant when compared with the river nutrient loading and total lake water content.

The objective of this phase of the research was to assess the significance of submerged aquatic macrophytes in White Lake. This inclues evaluation of the biomass and species distribution of the lake macrophytes, measurement and assessment of the significance of plant tissue nutrients to the total lake nutrient budget, and analysis of the potential of the macrophytes to respond to the sewage diversion project. A detailed discussion of these findings is presented in Appendix B.

Field and laboratory techniques were employed to investigate the importance of the submerged aquatic macrophytes in White Lake. The experimental methods are described in detail in Appendix B. Seven sampling cruises were conducted between June and October of 1974. The lake macrophyte community was mapped with respect to species, areal, and biomass distribution. Species distribution of the White Lake macrophyte community are illustrated in Figure 43. The species composition of the community in the late summer of 1974 is shown. At each keyed location all species identified are listed according to their relative abundance. Although a total of fourteen species were identified in the system, the relative abundance of species varied widely. *Myriophyllum verticillatum, Ceratophyllum demersum,* and *Vallisneria americana* were the prominent species at most locations, and represented the overwhelming majority of the biomass.

The results of macrophyte harvesting in White Lake indicate wide variability in plant densities both spatially and temporally with plant concentrations often approaching 150 g dry weight/meter^2 (typical for shallow eutrophic lakes). The observed extent of the macrophyte beds ranged from a 6% coverage of the lake bottom in June to a maximum coverage of 18% in August. The integrated results of the areal and biomass calculations derived from the seven macrophyte samplings are shown in Figure 44. The significance of these calculated biomass estimates can be evaluated if a comparison is made with the other major primary producer in the lake, the phytoplankton. Estimates of total chlorophyll *a* in the lake were made by multiplying the depth weighted average measured concentrations by the lake volume. An approximate stoichiometric dry weight to chlorophyll *a* ratio was used to estimate the phytoplankton biomass. A comparison of these results indicates that during much of the year the biomass standing crops of the macrophytes and the phytoplankton are approximately the same.

Samples of plant tissue were also analyzed for carbon, hydrogen, nitrogen, potassium, and phosphorus. These results suggest that the White Lake macrophytes community is not limited by phosphorus or nitrogen, and is probably restricted only by light and space requirements. Nutrients associated with the macrophytes are an order of magnitude smaller than the total nutrient content of the lake but are approximately one-half the magnitude of the estimated nutrient content of the phytoplankton. However, studies by Denny (1972) and calculations detailed in Appendix B indicate that macrophytes are capable of extracting significant quantities of phosphorus from the sediments. Because of this and other phenomena, it is important to consider the potential response of the community to the nutrient diversion program and reductions in lake levels. The possibility exists that the community will increase in size if light penetration of the water improves due to reduced levels of algae and detritus. In addition, lowered lake levels will increase the extent of the shallow water areas available for growth. Macrophytes can derive their essential nutrients from the rich bottom sediments; consequently, subsequent recycling of these nutrients to the overlying waters could result in the persistence of high nutrient and nuisance plankton levels. A program of direct macrophyte control may then be necessary.

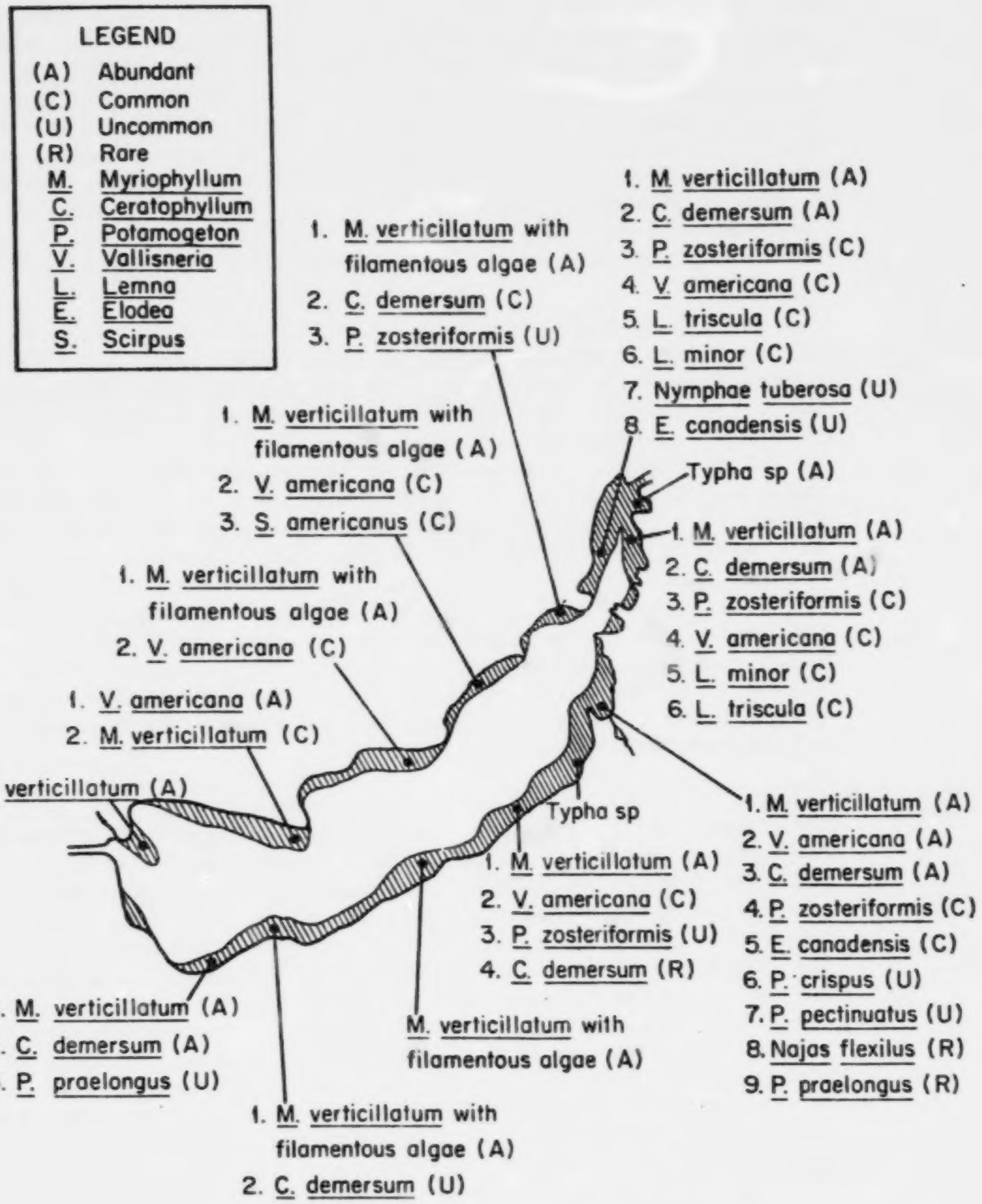

LEGEND
(A) Abundant
(C) Common
(U) Uncommon
(R) Rare
M. Myriophyllum
C. Ceratophyllum
P. Potamogeton
V. Vallisneria
L. Lemna
E. Elodea
S. Scirpus
1. M. verticillatum with filamentous algae (A)
2. C. demersum (C)
3. P. zosteriformis (U)
1. M. verticillatum (A)
2. C. demersum (A)
3. P. zosteriformis (C)
4. V. americana (C)
5. L. triscula (C)
6. L. minor (C)
7. Nymphae tuberosa (U)
8. E. canadensis (U)
Typha sp (A)
1. M. verticillatum (A)
2. C. demersum (A)
3. P. zosteriformis (C)
4. V. americana (C)
5. L. minor (C)
6. L. triscula (C)
1. M. verticillatum with filamentous algae (A)
2. V. americana (C)
3. S. americanus (C)
1. M. verticillatum with filamentous algae (A)
2. V. americana (C)
1. V. americana (A)
2. M. verticillatum (C)
M. verticillatum (A)
Typha sp
1. M. verticillatum (A)
2. V. americana (C)
3. P. zosteriformis (U)
4. C. demersum (R)
1. M. verticillatum (A)
2. V. americana (A)
3. C. demersum (A)
4. P. zosteriformis (C)
5. E. canadensis (C)
6. P. crispus (U)
7. P. pectinuatus (U)
8. Najas flexilus (R)
9. P. praelongus (R)
1. M. verticillatum (A)
2. C. demersum (A)
3. P. praelongus (U)
M. verticillatum with filamentous algae (A)
1. M. verticillatum with filamentous algae (A)
2. C. demersum (U)

Figure 43. Distribution of macrophytes in White Lake.

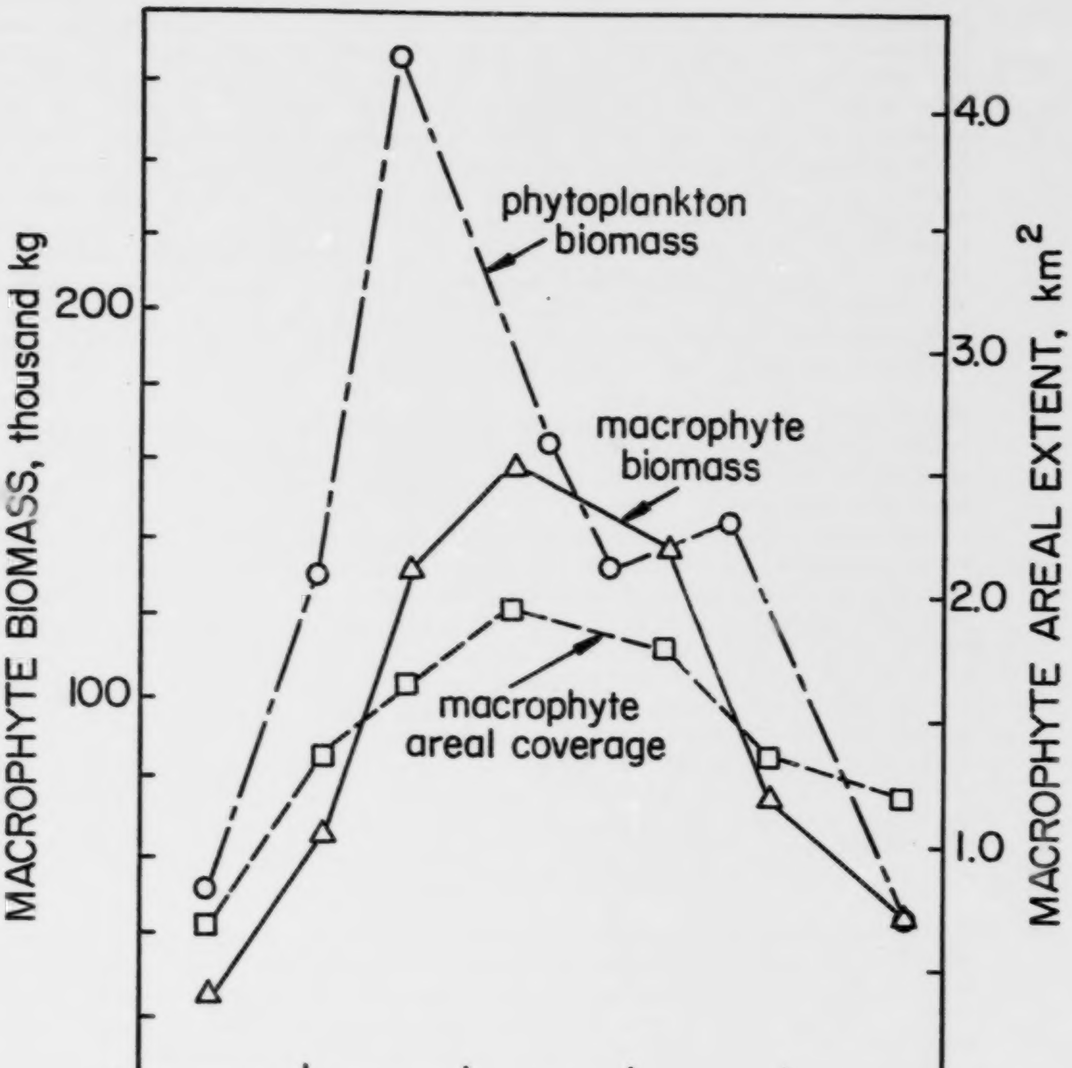


Figure 44. Macrophyte and phytoplankton content in White Lake, 1974.

Sediment-Nutrient Relationships -- The importance of lake sediments in controlling nutrient concentrations in overlying waters has long been a subject of concern. Mortimer (1941, 1942) concluded from his studies on English lakes that various mechanisms exist (e.g., adsorption, and complexation) in superficial oxidized sediment layers which may prevent the transport of materials from the sediments to overlying aerobic waters. More recently, it has been demonstrated that the disappearance of this oxidized microzone (as might occur during hypolimnetic anoxia) results in a substantial release of nutrients from the sediment (Mortimer, 1971). Harter (1968) found that eutrophic lake sediments have a large capacity to temporarily absorb phosphorus and later release it. In laboratory experiments, Porcella *et al.*, (1970) also demonstrated that sediments possess a large potential for nutrient release. Despite these studies, and numerous others, there does not now exist sufficient information to quantitatively predict the magnitude of nutrient release or uptake by sediments in White Lake (Lee, 1970, Byrnes *et al.*, 1972).

The purpose of the efforts summarized in this section was to conduct a study to assess the significance of sediment nutrient release under anaerobic conditions in White Lake. A detailed discussion of these findings is presented in Appendix C. A three-phase plan was devised to achieve the objectives of this study. The first task was to measure nutrient concentrations in the sediments and assess their potential for release and exchange with overlying waters. Investigations of the sediment revealed high nutrient concentrations in the interstitial waters. Results from these chemical analysis are shown in Figure 45. Examination of the data from the top five centimeters of sediments reveals significant concentration gradients between the sediments and the overlying waters. Maximum nutrient concentrations in this sediment zone were 22.0 mg Si/ℓ of silicon, 12.5 mg N/ℓ of ammonia, 3.6 mg P/ℓ soluble reactive phosphorus, and 5.2 mg P/ℓ total dissolved phosphorus. The calculated concentration gradients between the sediment and overlying waters were 6.1 mg Si/ℓ/cm silicon, 3.4 mg N/ℓ/cm ammonia, 2.36 mg P/ℓ/cm soluble reactive phosphorus, and 3.4 mg/ℓ/cm total dissolved phosphorus.

The second phase of the work involved a laboratory study to simulate the lake bottom and measure the sediment-water nutrient flux. The experiments were conducted on sediments and hypolimnetic water collected in a benthic corer. These experiments revealed significant nutrient release from the sediments to the overlying waters under anaerobic conditions. The results from this experiment are shown in Figure 46. Observed concentrations of nutrients in the overlying waters increased drastically. Soluble reactive phosphorus concentrations increased from 0.05 to 0.56 mg P/ℓ, while total dissolved phosphorus increased from 0.05 to 0.67 mg P/ℓ. Ammonia increased from 0.23 to 0.78 mg N/ℓ and silicon increased from 2.2 to 8.7 mg Si/ℓ. Nutrient flux rates were calculated from nutrient release data by use of graphical and analytical techniques incorporating the area and volume of the experimental apparatus.

The third phase of this work consisted of measurements of actual nutrient release rates in a chamber placed at the bottom of White Lake. The results from the *in situ* chamber study also revealed high sediment-water

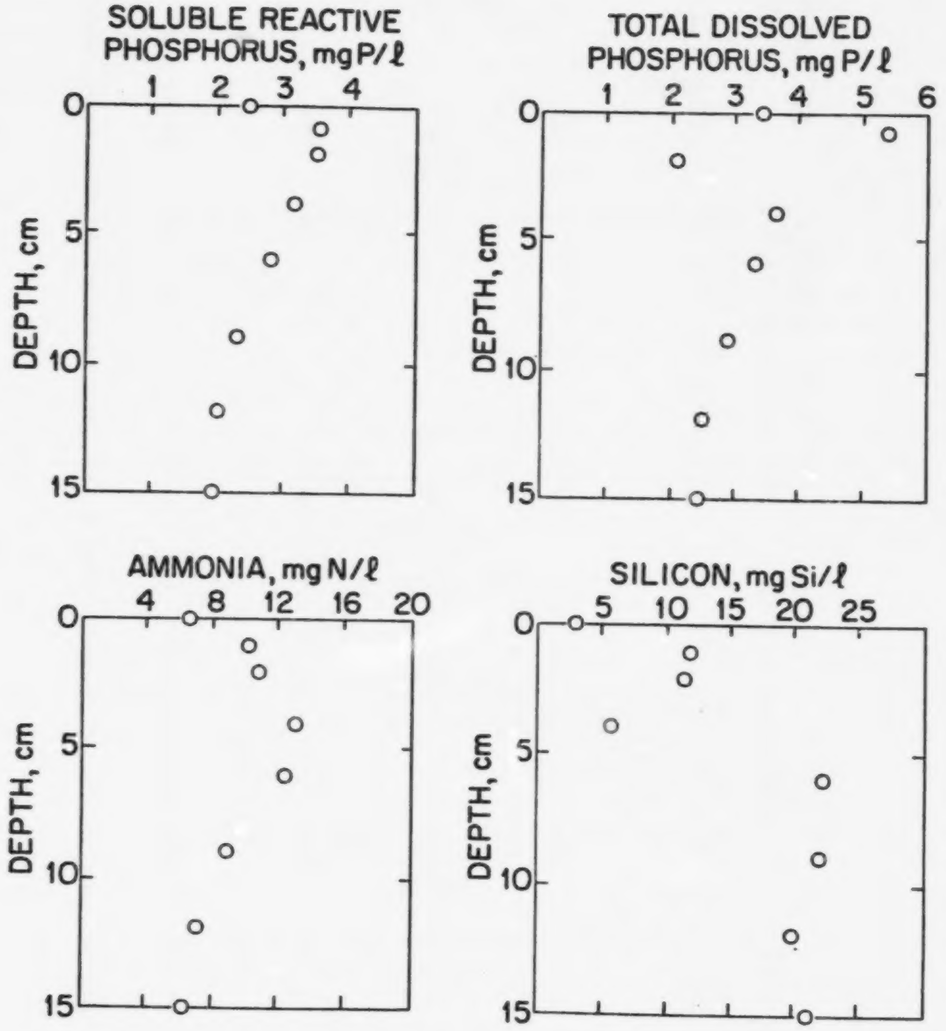


Figure 45. Concentrations of nutrients in White Lake sediment interstitial waters.

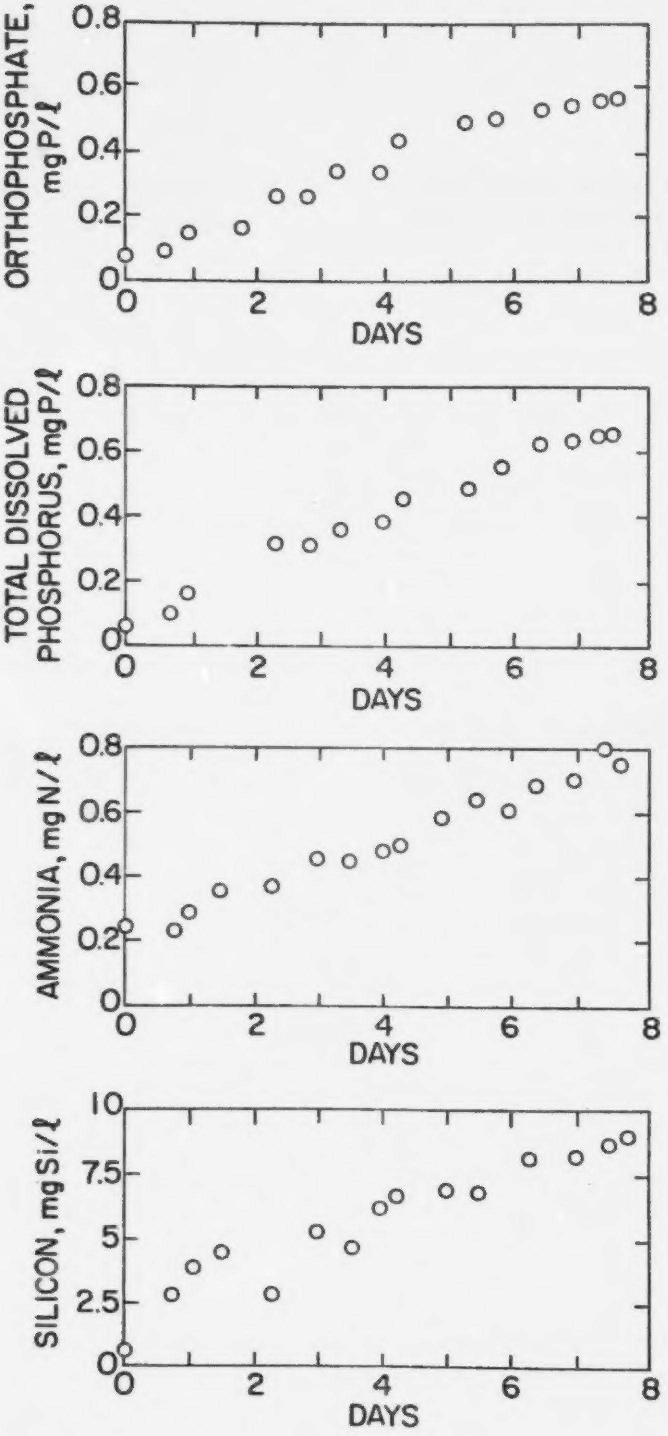


Figure 46. Nutrient release from White Lake sediments-laboratory studies.

nutrient exchange rates. A plot of the change of nutrient concentrations in the chamber is shown in Figure 47. Concentrations of soluble reactive phosphorus increased from 0.04 to 0.20 mg P/ℓ during the three-day study, ammonia increased from 0.33 to 0.61 mg N/ℓ, and silicon increased from 1.1 to 4.5 mg Si/ℓ.

The results of these studies were used to calculate a flux rate between the sediment and water. General agreement among methods was obtained for both nitrogen and phosphorus. The average sediment nutrient release rates for the three methods were 25.1 mg P/m^2/day soluble reactive phosphorus and 36.2 mg N/m^2/day ammonia. The associated net diffusion coefficient across the sediment-water interface was calculated to be 1.0 to 1.5 x 10^{-5} cm^2/sec. Theoretical and experimental results for silicon release were not in good agreement. Speculations regarding this discrepancy are discussed in Appendix C.

The measured flux rates of nitrogen and phosphorus suggest that the sediments have the potential to contribute nutrients to the overlying waters at levels sufficient to support algal growth. As a result, improvements in water quality resulting from the White Lake sewage diversion project could be delayed or reduced.

Modeling -- One goal of the Muskegon Wastewater Management System is to improve the water quality of White, Muskegon, and Mona Lakes so that they may be more effectively used for recreation and navigation. This usage is a function of the levels of algae, macrophytes, dissolved oxygen, and fecal coliform in the lake. The limnological sampling program has been conducted to document changes in water quality following land disposal treatment. Data from this sampling program has also been used to develop mathematical models which can be used to predict the long-term impact of the waste management program. Mathematical models are important because they allow a quantitative evaluation of the relative importance of several factors which may effect water quality in lakes. For example, algal blooms in lakes are a function of nutrient loading from non-point sources such as urban and agricultural runoff, domestic and industrial wastes, nutrients released from the bottom sediments, nutrients pumped into the water by rooted aquatic macrophytes, and the rate of nutrient recycle in the water column. All of these processes must be quantitatively evaluated before improvements in levels of phytoplankton productivity and other parameters related to water quality can be predicted following a decrease in nutrient loading or implementation of other restorative measures such as sediment dredging and macrophyte harvesting. Experience in Lakes Washington and Sammamish has shown that the effectiveness of nutrient diversion cannot always be intuitively predicted and that mathematical models can explain otherwise unexpected results.

The goal of the modeling studies was to predict long-term phosphorus concentrations in White Lake. Most previous phosphorus and trophic status models (Vollenweider, 1969, Vollenweider, 1975, and Dillon and Rigler, 1974) have considered a single component of the phosphorus cycle (total phosphorus). Most of these models have been applied to less eutrophic waters for cases where sediments play a minor role in regulating

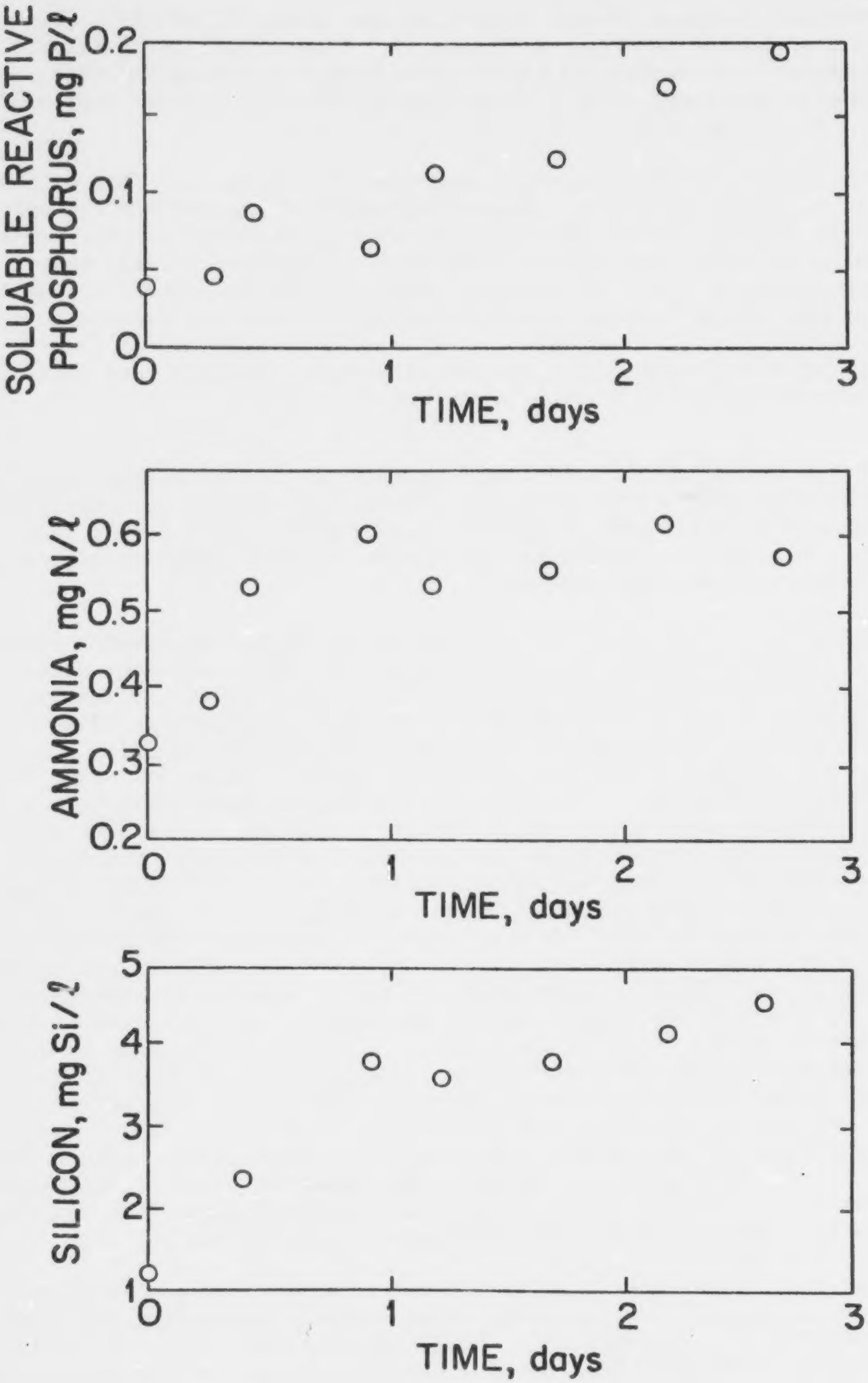


Figure 47. Nutrient release from White Lake sediments-*in situ* studies.

the phosphorus cycle. Under many circumstances, such as in White Lake, other phenomena, such as phosphorus regeneration from the sediments to the overlying water and phosphorus transformation within the sediments may complicate the system response and require more comprehensive models. Therefore, a model which incorporates the water and the sediment systems and considers two forms of phosphorus (particulate and dissolved) has been developed for White Lake. A significant amount of phosphorus release has been observed in the hypolimnion of the lake during anaerobic conditions in the summer. The model quantifies the dynamic interactions of phosphorus between sediments and water by taking into account the sinking of particulate phosphorus and diffusion of dissolved phosphorus across the sediment-water interface. Other model mechanisms include vertical eddy diffusion in the water, phosphorus transformation between the particulate form and the dissolved form in both the water and the sediment, diffusion of phosphorus in the interstitial water, and sedimentation in the sediments. The components of the model are displayed in Figue 48.

Extensive field data were used to determine the coefficients and parameters defined in the model formulations. The vertical eddy diffusion coefficient characterizing vertical mixing between the epilimnion and the hypolimnion was calculated using temperature and nutrient conditions. The sinking velocity of particulate phosphorus in the lake was determined through mass balance considerations. The diffusion coefficient in the interstitial water was determined from the chloride profile in the sediment. The kinetic coefficient in the upper sediment layers was found to correlate with the seasonally variable dissolved oxygen level at the bottom of the lake. A detailed discussion of the model formulation is contained in Appendix D.

Model calibration produced close agreement between the model calculations and observed data for 1974 (see Figure 49). Sensitivity analysis further substantiated the model results. The model was then verified against an independent set of data taken in 1973 with the parameters and coefficients determined in the 1974 calibrations. The model is able to explain the significant release of phosphorus from the sediment to the hypolimnion of White Lake in summer. Inclusion of two separate forms of phosphorus into the model was necessary to gain detailed insight into the dynamics of phosphorus cycling in White Lake.

One immediate application of the model was to assess the effect of nutrient reduction resulting from the sewage diversion program on White Lake. The simulation results shown in Figure 50 indicate a slight decrease of phosphorus levels in the first year after nutrient reduction and virtually no improvement after that. It is concluded, therefore, that the lake will not undergo dramatic improvement in water quality (i.e., reduction

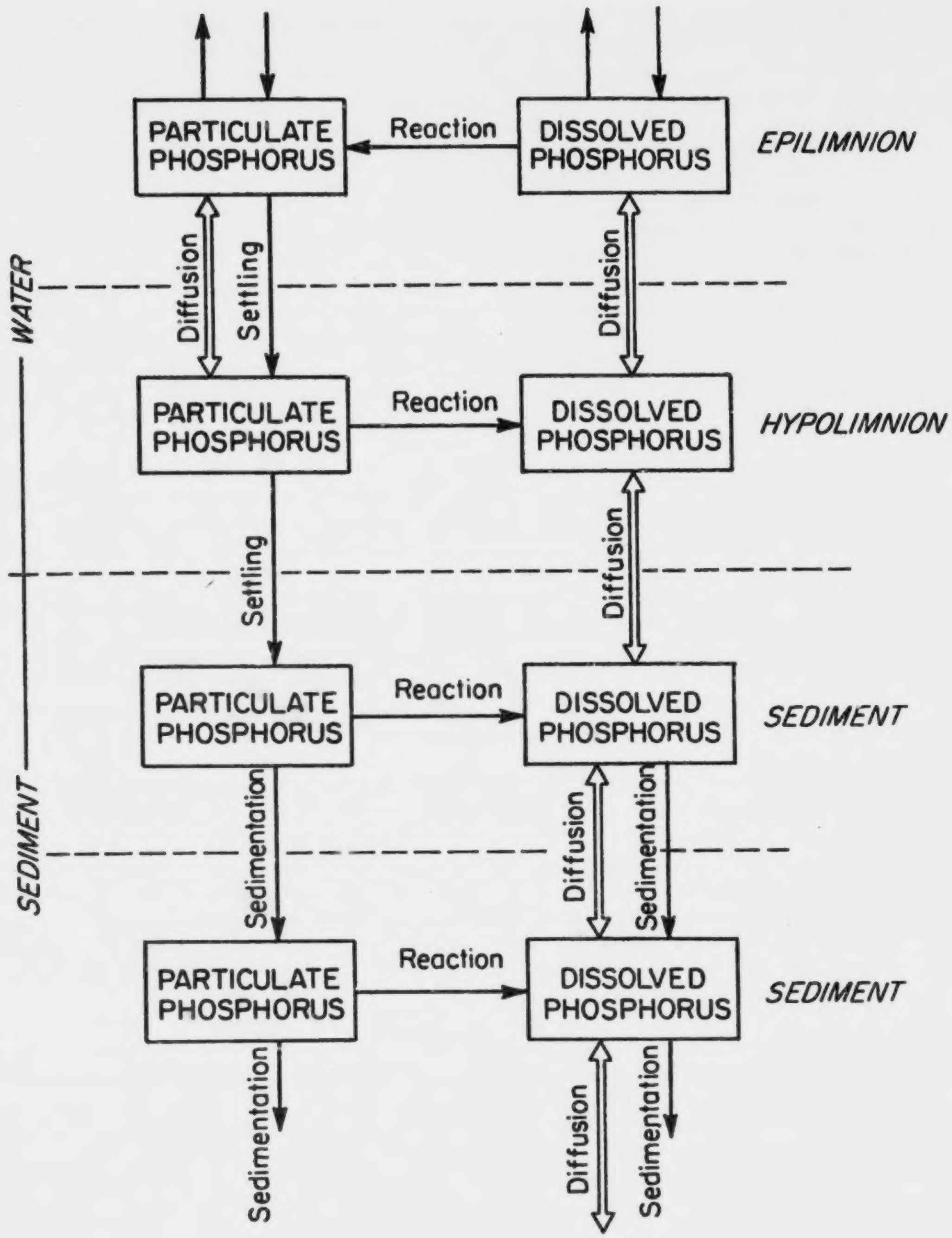

PARTICULATE PHOSPHORUS
Reaction
DISSOLVED PHOSPHORUS
EPILIMNION
Diffusion
Settling
Diffusion
WATER
PARTICULATE PHOSPHORUS
Reaction
DISSOLVED PHOSPHORUS
HYPOLIMNION
Settling
Diffusion
PARTICULATE PHOSPHORUS
Reaction
DISSOLVED PHOSPHORUS
SEDIMENT
Sedimentation
Diffusion
Sedimentation
SEDIMENT
PARTICULATE PHOSPHORUS
Reaction
DISSOLVED PHOSPHORUS
SEDIMENT
Sedimentation
Diffusion
Sedimentation

Figure 48. White Lake phosphorus model.

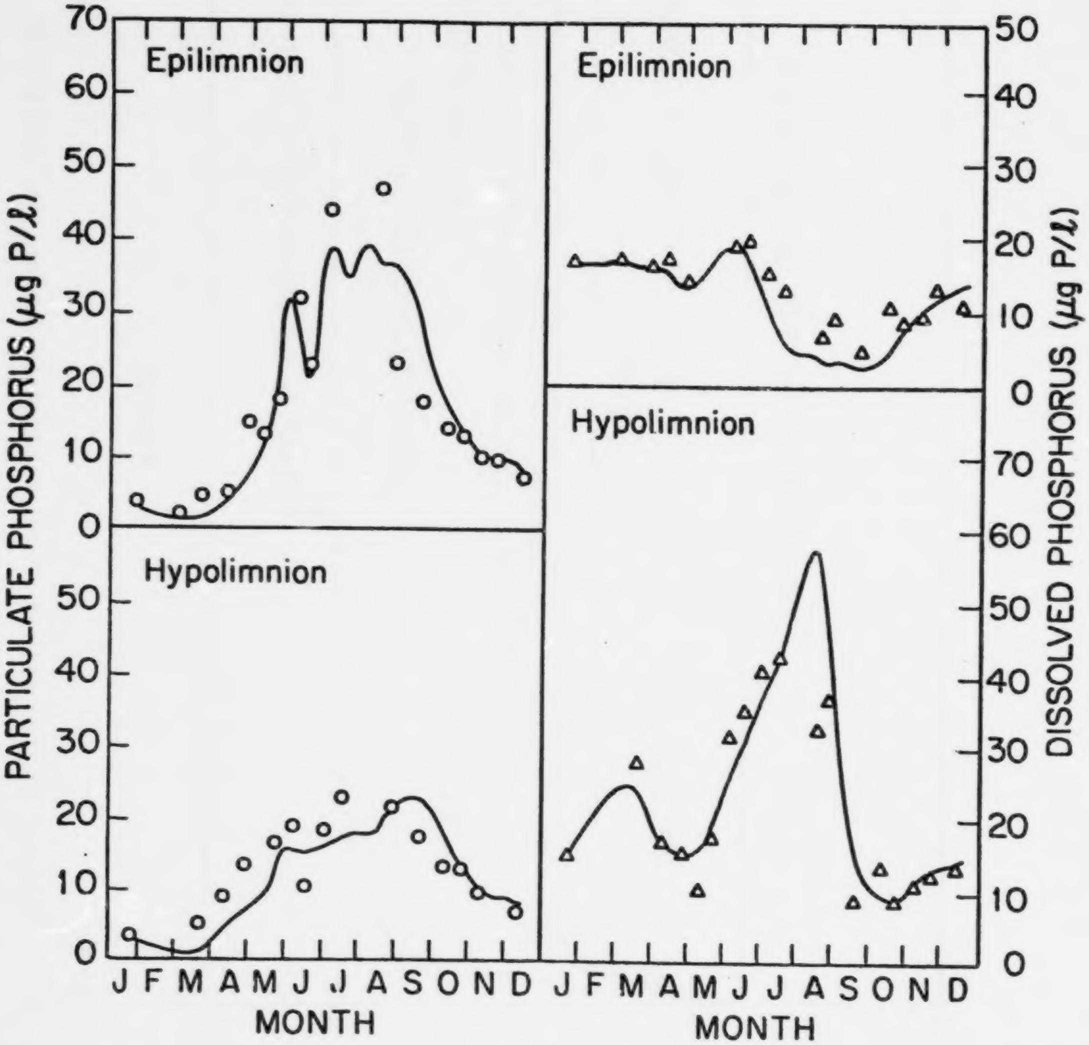


Figure 49. Comparison of model calculations and observed data (1974) for dissolved and particulate phosphorus in White Lake.

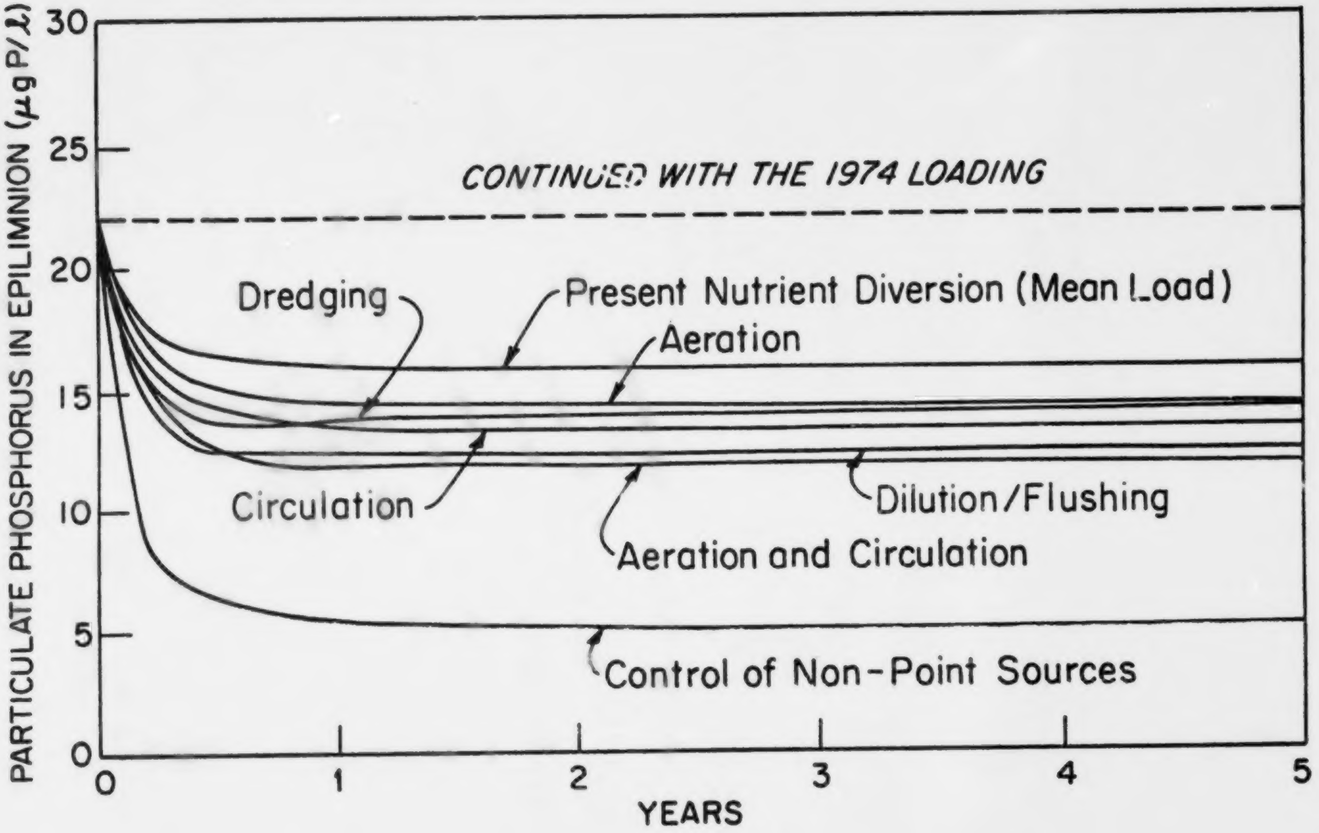


Figure 50. Comparison of rehabilitation schemes for White Lake.

in algal concentrations). The model was also used to evaluate the additional schemes of lake rehabilitation for the possibility of further improvement of water quality in White Lake. Figure 50 demonstrates that further reduction of the present non-point phosphorus loading is necessary for addition recovery, although some improvements would also be expected to result from aeration, mixing, dredging, and flow augmentation.

SECTION IX

MUSKEGON LAKE

INTRODUCTION

Muskegon Lake is the largest of the three study lakes (see Figure 3). The lake has a surface area of 16.8 km^2, a volume of 12.0 x 10^7 m^3 and a mean depth of 7.1 meters. The maximum lake depth is 23 meters. There exist shallow littoral areas in the lake; however, these are not as extensive or as over-grown by macrophytes as those of White Lake. The lake has a roughly east-west orientation and is approximately 8 kilometers long by 1.5 to 3.5 kilometers wide.

Muskegon Lake receives flow from six identified tributaries of which the Muskegon River and Bear Lake Creek are the most significant. The lake discharges into Lake Michigan through a narrow channel. The channel and the lake are dredged permitting the commercial navigation important to various industries.

The area surrounding Muskegon Lake is mixed urban, industrial, and residential. The major segment of the industry and the urban center lies to the southeast and east of the lake. The southwest and north shores are primarily residential. In the past, the lake has received numerous industrial and municipal discharges which are described in detail in subsequent sections. The most notable of these discharges were from the Muskegon and North Muskegon Sewage Treatment Plants, the S.D. Warren Paper Company, and the Storey (Ott) Chemical Company.

TRIBUTARY-RELATED CONSIDERATIONS

Hydrology -- The Muskegon Lake drainage basin has an area of 6,819 km^2 and drains lands from Muskegon, Newaygo, Montcalm, and Mecosta Counties. Six identified tributaries empty into Muskegon Lake, with the Muskegon River dominating the system (see Figures 5 and 7). The Muskegon River drains 97.5% of the Muskegon Lake basin with the next largest tributary Bear Lake Creek, accounting for only 1.1% (see Table 18). Hydrologically the Muskegon River is also dominant. In a 1972 to 1973 study by the USEPA (1975c) the Muskegon River was observed to contribute an estimated 95% of the total flow to Muskegon Lake.

The Muskegon River basin begins in northern Mecosta County and flows southwesterly to Lake Michigan. Because the focus of this study is on Muskegon Lake, all references to the Muskegon River and its basin will be with regards to the drainage upstream of the river entry to Muskegon Lake. Tributary and direct drainage to the lake will be considered

Table 18. MUSKEGON LAKE DRAINAGE BASIN CHARACTERISTICS

Description	Area (km^2)	Percentage of Total	Discharge (m^3/sec)	Percentage of Total
Muskegon River	6645.7	97.5%	58.86	97.00%
Green Creek	30.3	0.4%	0.33	0.54%
Bear Lake Creek	74.0	1.1%	0.84	1.38%
Four Mile Creek	8.8	0.1%	0.09	0.15%
Ryerson Creek	19.7	0.3%	0.20	0.33%
Ruddiman Creek	8.8	0.1%	0.09	0.15%
Minor Tributaries & Immediate Drainage	15.0	0.2%	0.27	0.44%
Muskegon Lake	16.8	0.2%		
Total	6819.1	100%	60.68	100%

(Modified from USEPA, 1975c).

separately. The 52-year average flow in the Muskegon River as measured at USGS station 04122000 at Newaygo is 54.9 m^3/sec. The annual average flow can vary significantly (see Figure 51). Between 1972 and 1973 the river changed from a moderately low flow period to a high flow period. The monthly average flows as measured at the USGS station during this study are shown in Figure 52.

The flow at the USGS station does not represent the total flow of the Muskegon River to Muskegon Lake. Between the USGS station and the river mouth the Muskegon River receives drainage from six identified tributaries (Cedar Creek, Mosquito Creek, Brooks Creek, Maple River, Minnie Creek, and Sand Creek), several minor tributaries, and direct drainage sources. To estimate the total river flow entering Muskegon Lake, the USGS flows were multiplied by 1.23. This factor was derived from miscellaneous USGS flow data (USEPA, 1975c), flow data collected during this study, and drainage area considerations. The total 52-year average flow entering Muskegon Lake is then calculated to be 67.5 m^3/sec. Direct municipal and industrial discharges do not significantly alter the hydrologic characteristics of the river.

The significance of the Muskegon River flow to Muskegon Lake can be calculated if the ratio of lake volume to tributary flow or mean hydraulic retention time is calculated. The lake volume is 12.0×10^7 m^3. The retention time is calculated to be 21 days. This indicates that on the average the Muskegon River has the capability of flushing the total volume out of the lake over 16 times per year. This hydraulic retention time is, of course, an average estimate which does not consider short-circuiting, stratification, etc., nor does it consider the seasonal variation in flow which would cause the retention time to vary at different times of the year.

Concentrations of Chemical Species in Tributaries -- Measurements of various water quality parameters were made in the Muskegon River and Bear Lake Channel. Only these tributaries were chosen for monitoring. Hydrologic evaluations demonstrated that all other tributaries were insignificant and relatively uncontaminated. Bear Lake Channel was assumed representative of general water quality in the minor tributaries. A summary of the water quality measurements in the Muskegon River and Bear Lake Channel is given in Table 19. The Muskegon River discharge concentrations displayed in this table represent an average of the concentrations measured in the three branches of the river. Each of the branches originate from interconnected sources and the relative proportion of flows in each branch could not be distinguished.

A comparison of the concentrations of chemical species at the Muskegon River mouth with those in the Bear Lake Channel show the two to be of similar magnitude except for chloride and nitrogen. The Bear Lake Channel

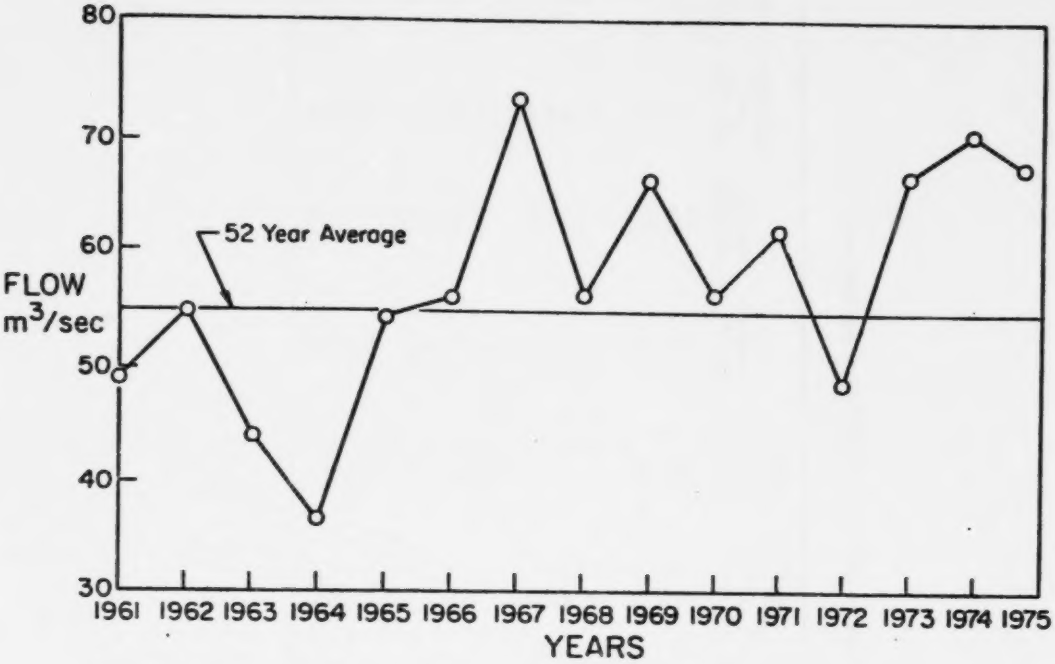


Figure 51. Muskegon River average annual flow at Newaygo, Michigan (USGS Station #04122000).

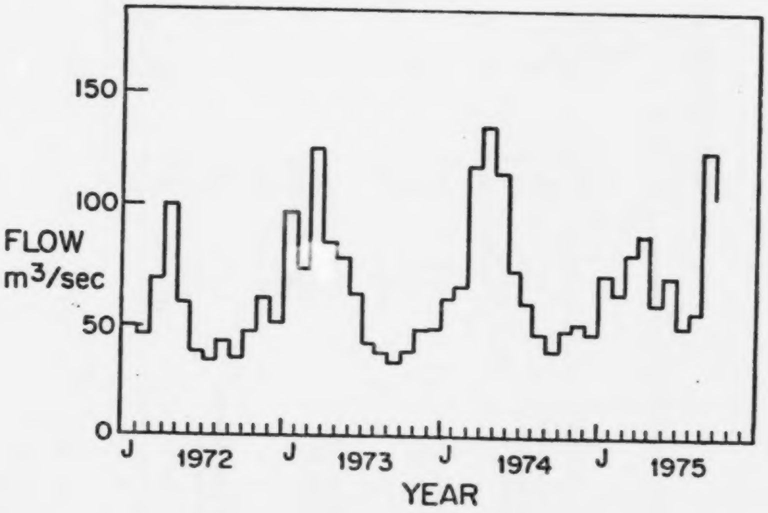


Figure 52. Muskegon River mean monthly flow at Newaygo, Michigan (USGS Station #04122000).

Table 19. AVERAGE CONCENTRATIONS OF SELECTED CHEMICAL SPECIES IN TRIBUTARIES TO MUSKEGON LAKE, 1972-1975

Tributary	Year	Chloride (mg Cl/ℓ)	Ammonia (μg N/ℓ)	Nitrate (μg N/ℓ)	Soluble Reactive Phosphorus (μg P/ℓ)	Total Dissolved Phosphorus (μg P/ℓ)	Total Phosphorus (μg P/ℓ)	Silicon (mg Si/ℓ)	Dissolved Iron (μg Fe/ℓ)	Total Iron (μg Fe/ℓ)
Muskegon River (average of all branches)	1973	19.0	85	101	18	40	63	2.3	76	485
	1974	20.9	35	151	11	22	46	2.6	63	149
	1975	21.5	33	200	9	23	43	3.1	57	157
	1973-1975	20.3	54	144	13	29	51	2.6	66	260
Muskegon River (Station 143)	1973	18.6	17	102	5	15	32	2.2	61	461
	1974	18.4	30	163	6	13	27	2.5	41	111
	1975	18.9	24	211	5	13	33	2.9	33	124
	1973-1975	18.6	24	156	5	14	30	2.6	44	220
Muskegon River (Station 102)	1972	15.4	57	409	--	18	29	3.1	--	--
	1973	17.9	18	97	14	30	36	2.7	64	454
	1974	25.5	45	174	14	22	41	2.7	70	170
	1975	27.7	31	228	13	26	67	3.1	79	148
	1972-1975	22.0	35	196	14	25	44	2.8	71	250
Bear Lake Channel (Station 148)	1973	43.1	42	50	4	14	70	2.1	52	310
	1974	39.7	37	90	3	12	43	1.7	49	146
	1975	38.6	41	91	4	17	60	3.1	70	154
	1973-1975	40.6	40	77	3	14	57	2.3	56	194

chlorides were approximately twice those in the Muskegon River. This elevation in chloride concentrations is postulated to be an effect of the high salt content of the groundwater inflow. No other explanation is proposed. No point discharges are known to exist in the Bear Lake basin. Nitrate and ammonia concentrations in the Bear Lake Channel were 30-50% less than those in the Muskegon River. This is considered a consequence of point discharges to Muskegon River waters. Since the concentrations of chemical species in the Bear Lake Channel were not excessive and because of its hydrologic insignificance, all subsequent discussions of water quality are focused on Muskegon River.

The Muskegon River is the dominant tributary to Muskegon Lake and thus the lake water quality can be expected to be a function of tributary water quality. This assumption ignores any direct point discharges to the lake which could change concentrations. The average river concentrations of dissolved inorganic nitrogen (199 μg N/ℓ), total phosphorus (52 μg P/ℓ), and dissolved silicon (2.6 mg Si/ℓ) are capable of supporting significant algal growth. Although only 25% of the total phosphorus is in the readily available soluble reactive form, all of the phosphorus is expected to be potentially available through bacterial and enzymatic action.

Figures 53 to 58 are plots of concentrations of various parameters measured in the Muskegon River. The seasonal patterns observed are similar to those found in the White River. Higher phosphorus concentrations were found in the late spring and summer and are associated with higher runoff while higher nitrate concentrations were measured during the winter and reflect low plant uptake.

Yearly average concentrations of chemical species measured in the Muskegon River display a significant variability. Explanations for this variation must include analysis of upstream concentrations as they relate to downstream water quality. This isolates the effects of the diversion system from changes in upstream boundary condition influence. Concentrations of plant nutrients measured at the river mouth reflect upstream water quality (including spray drainage) during all years except 1973. During 1973 observed ammonia and phosphorus concentrations were higher at the river mouth, a result of industrial and municipal discharges. In 1972 no measurements were made at the river mouth. The two most significant discharges to the river, the Muskegon Wastewater Treatment Plant and the Storey (Ott) Chemical Company, diverted their wastes in May 1973 and April 1974, respectively.

A reduction of the ammonia and phosphorus concentrations at the Muskegon River mouth was observed after 1973. Average yearly concentrations between 1973 and 1974 dropped from 62.7 μg P/ℓ to 45.8 μg P/ℓ for total phosphorus and 85.5 μg N/ℓ to 35.4 μg N/ℓ for ammonia. Total iron also showed a decrease from 485 μg Fe/ℓ to 149 μg Fe/ℓ. Other parameters did not significantly change. The river mouth data were compared

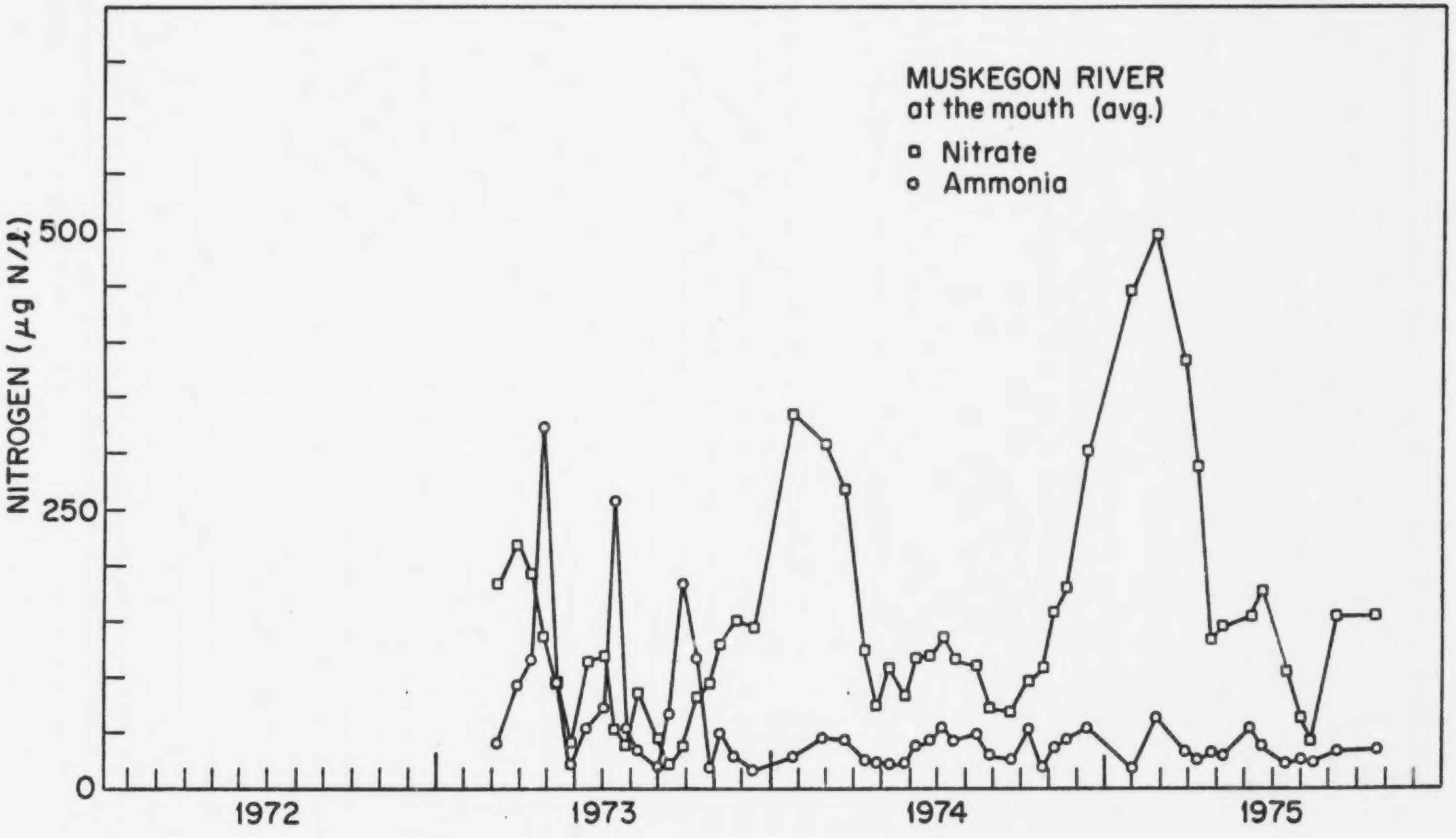


Figure 53. Dissolved inorganic nitrogen concentrations in the Muskegon River, 1973-1975.

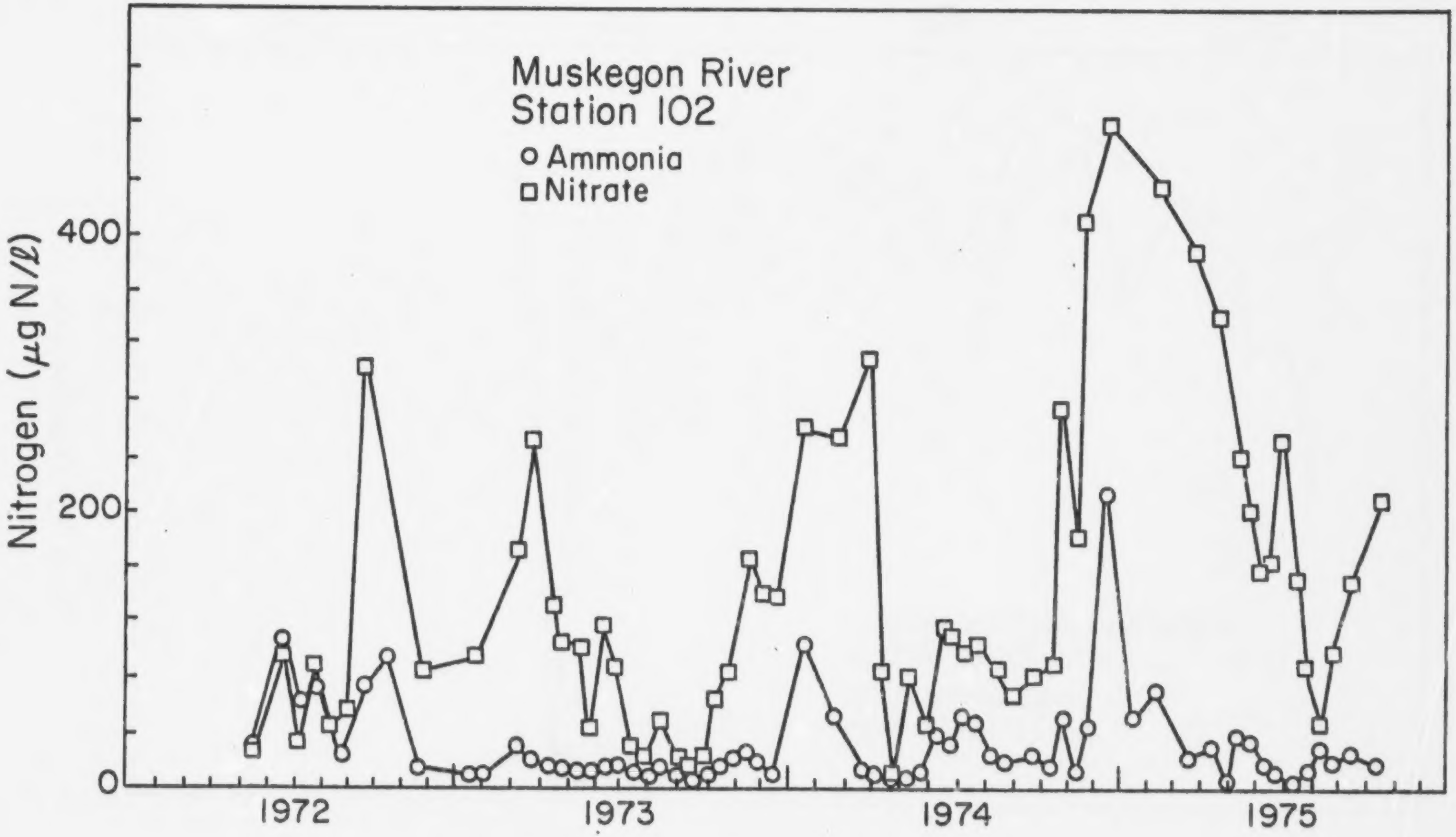


Figure 54. Dissolved inorganic nitrogen concentrations in the Muskegon River, 1972-1975.

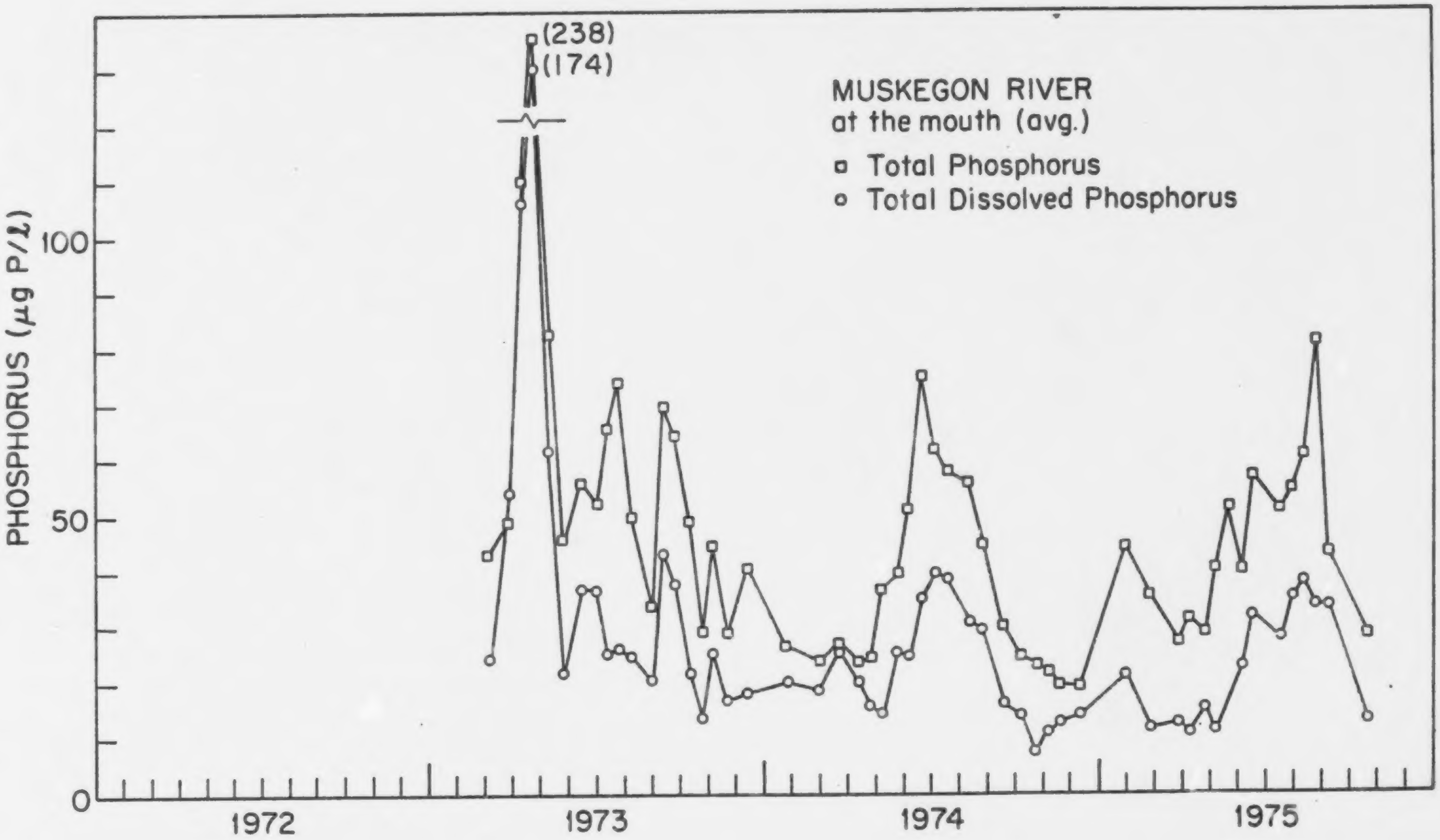


Figure 55. Phosphorus concentrations in the Muskegon River, 1972-1975.

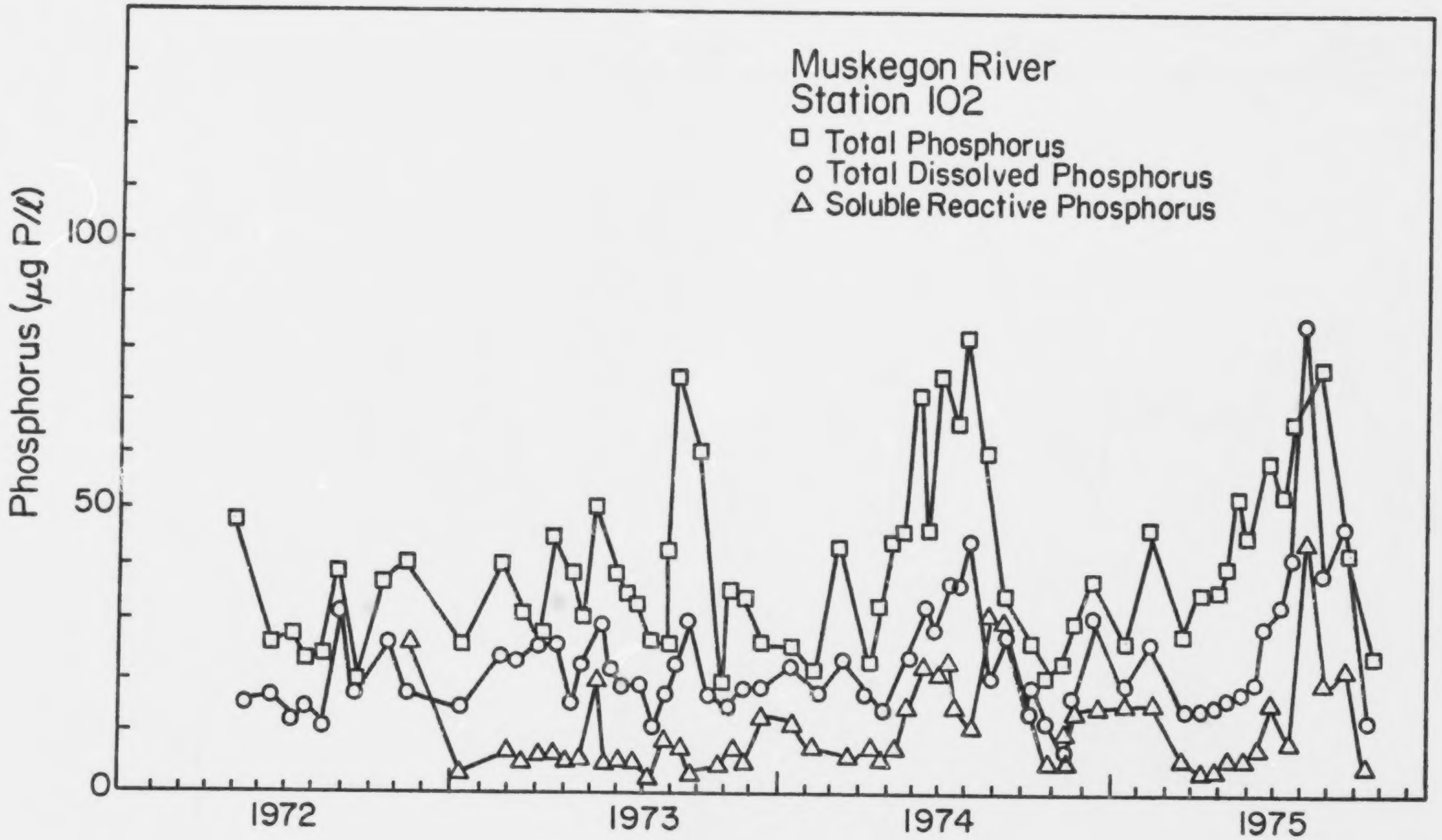


Figure 56. Phosphorus concentrations in the Muskegon River, 1972-1975.

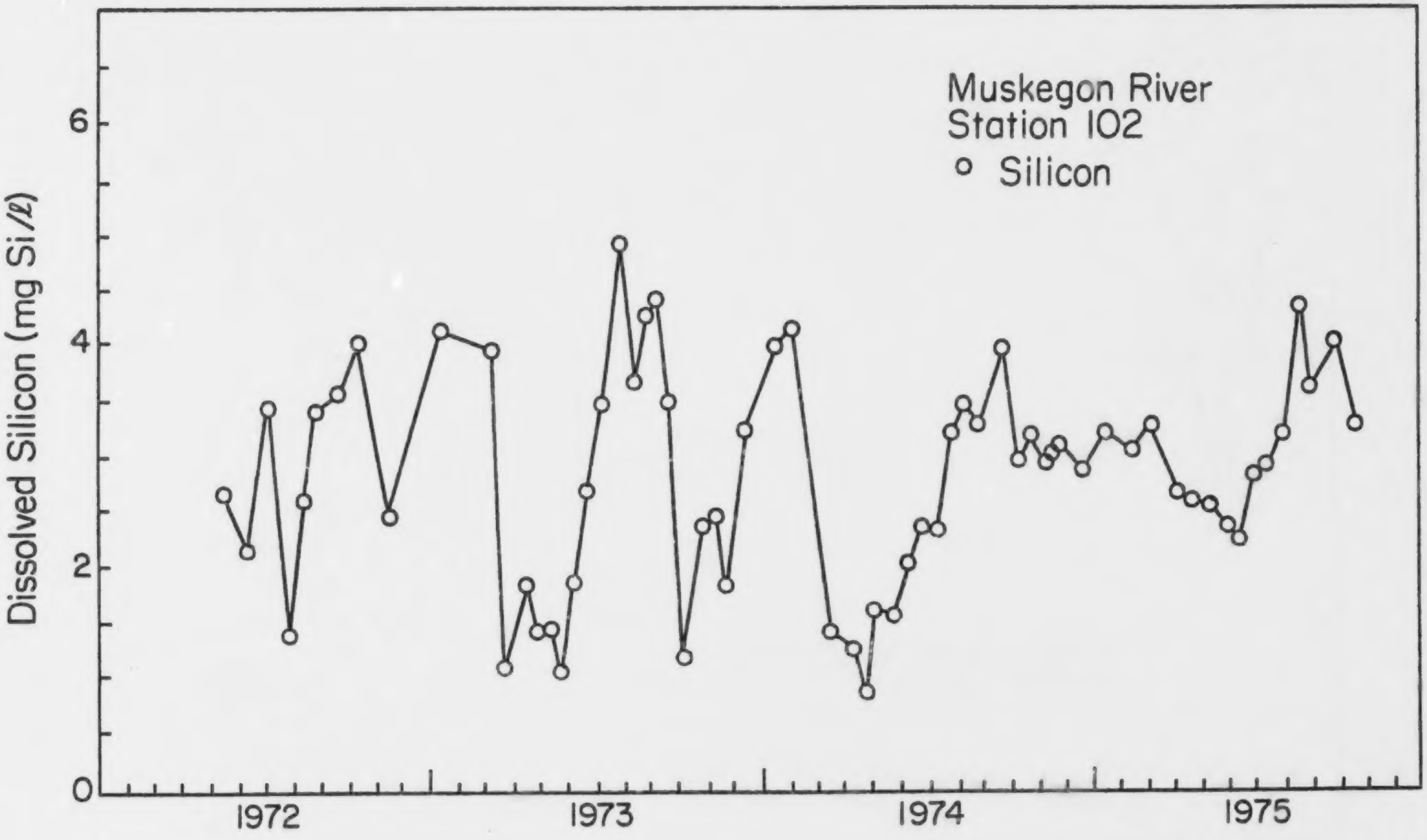


Figure 57. Silicon concentrations in the Muskegon River, 1972-1975.

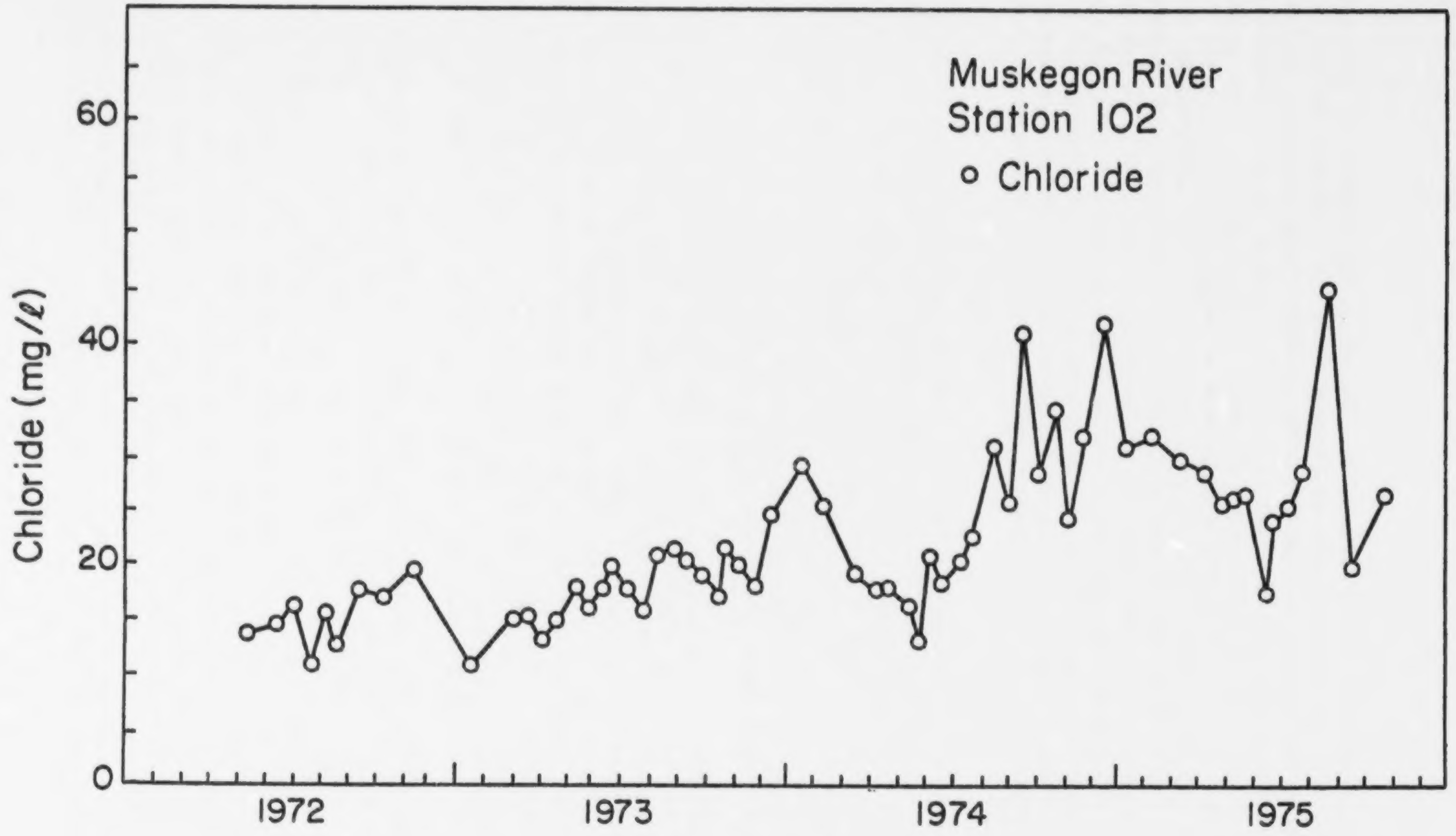


Figure 58. Chloride concentrations in the Muskegon River, 1972-1975.

to upstream data from Station 102. Prior to the summer of 1973 the Muskegon River received the discharges from the Muskegon and North Muskegon Wastewater Treatment Plants between these stations, and hence a comparison can relate cause and effect. A drop in iron concentrations between 1973 and 1974 was also observed at Station 102. This suggests that the decrease in concentration was a consequence of upstream conditions not diversion. The decrease in iron was also observed at Station 143 which is upstream of the spray site; this rules out any cause-and-effect relation to the spray diversion project for iron. On the other hand, the phosphorus and ammonia data suggest that the observed reductions for these parameters may have been a direct consequence of the waste diversion because similar decreases in phosphorus and ammonia concentrations were not observed at upstream stations between the years of 1973 and 1974.

The Storey (Ott) Chemical Company diverted high ammonia containing wastes in early 1974. No subsequent significant reduction in ammonia was observed subsequent to diversion. Part of the observed reduction in 1973 and 1974 in ammonia could have been caused by this diversion. However, as will be noted later, the municipal waste discharge loads were much greater than those originating from Storey (Ott) Chemical Company. It is difficult therefore to directly distinguish the effect of the Storey (Ott) Chemical Company dishcarge.

Limited effects on Muskegon River concentrations were observed to result from the drainage from the spray-irrigation site. This can be evaluated by examining the yearly averages at Stations 143 and 102. An increase in nitrogen concentrations might have been expected to result from the application of urea fertilizer to the spray site. This was not observed. Either this effect was minimal or the sampling dates did not coincide with fertilization periods. No specific information was available on the amounts and dates of fertilization. Total phosphorus concentrations were observed to increase between Stations 143 and 102 from 26.7 to 41.4 μg P/ℓ in 1974 and 32.6 μg P/ℓ to 67.4 μg P/ℓ in 1975.

During 1975, the average concentrations of nitrogen and phosphorus measured in Mosquito Creek (which receives the drainage from the spray site) were 50 μg P/ℓ total phosphorus, 1.9 mg N/ℓ nitrate, and 0.6 mg N/ℓ ammonia (Muskegon County, 1976). Concentrations measured in other years were of similar magnitude or less (see Appendix E). The total phosphorus concentrations were not significantly different than upstream Muskegon River concentrations. This illustrates the effectiveness of the soil to adsorb phosphorus from percolating water. Nitrate and ammonia, on the other hand, were significantly higher in the drainage than in upstream locations. The potential effect of the irrigation drainage on the Muskegon River can be evaluated by comparing the spray drainage flow and the Muskegon River flow. The average 1975 drainage from the spray site was approximately 1 m^3/sec which represents less than 2% of the total Muskegon River flow. Given this consideration and the similarity between drainage phosphorus and river phosphorus concentrations, the observed increase in phosphorus between Stations 143 and 102 cannot be readily explained.

The distribution of phosphorus among its various forms was noted to have changed during the investigation. During 1973, solible reactive phosphorus concentration represented approximately 33% of the total phosphorus concentration at the river mouth. During subsequent years this fraction dropped to about 25%. The percentage may have been higher prior to 1973 when industrial discharges were at their maximum; however, no information is available on this aspect. The point industrial discharges of phosphorus, particularly from the wastewater treatment plant, are principally in the soluble reactive form, hence the trend was as expected. Phosphorus data also showed the soluble reactive form of phosphorus to increase in percentage with distance downstream. This was particularly evident between Stations 143 and 102. This could be an effect of differences in non-point loads or due to bacterial and enzymatic conversion of organic forms of phosphorus to the soluble reactive form.

Dissolved silicon was noted to decrease in concentration at the river mouth as compared to upstream levels. This could be an effect of algal uptake or dilution from intruding Muskegon Lake waters. Average chloride concentrations were observed to increase downstream of the spray site (Station 102) as compared to Station 143. This increase did not appear in the Muskegon River mouth data. Higher chlorides might be expected from the spray drainage; however, the downstream disappearance of this increase is inconsistent.

In general, the wastewater diversion project has resulted in improvements in Muskegon River water quality. Prior to diversion, the industrial and municipal discharges caused elevated levels of phosphorus and ammonia in the river. The effect of diversion could have been portrayed more clearly if additional pre-diversion river mouth data were available. The spray site drainage did not appear to have a deleterious influence on Muskegon River water quality. However, variation in upstream concentrations resulting from non-point sources is a dominant factor in determining nutrient concentrations in the Muskegon River during the post-diversion period.

Nutrient Loads -- The major sources of nutrients to Muskegon Lake have been from municipal, industrial, and tributary discharges (see Figure 59). Other loads are described in Appendix F. Six tributaries empty into Muskegon Lake; of these the Muskegon River is the only significant source of nutrients. A 1972-73 study by the USEPA (1975c) indicated that the Muskegon River accounted for 94% of the total phosphorus load from the tributaries and 96% of the total nitrogen load.

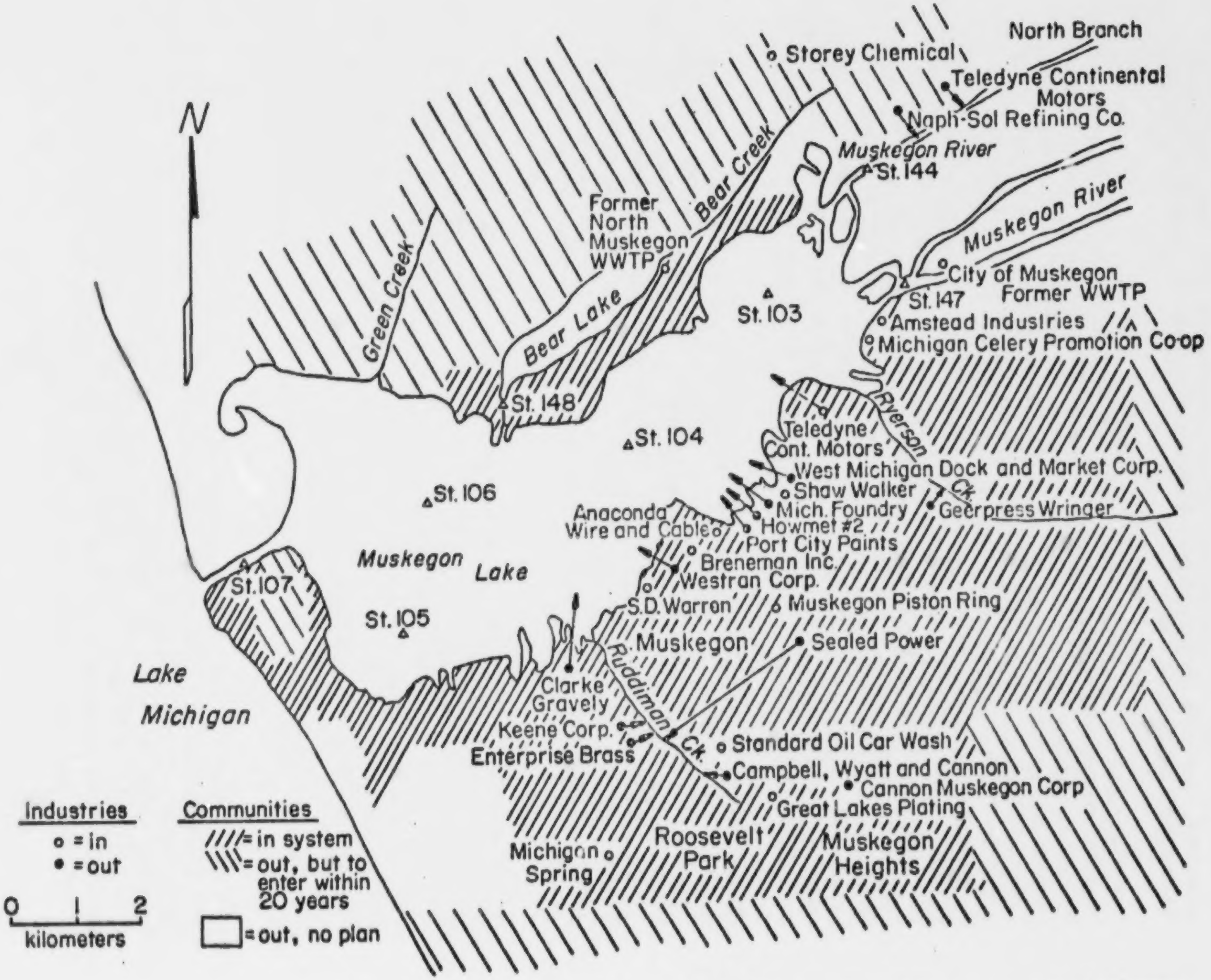

N
North Branch
Storey Chemical
Teledyne Continental Motors
Naph-Sol Refining Co.
Muskegon River
St. 144
Muskegon River
City of Muskegon Former WWTP
St. 147
Amstead Industries
Michigan Celery Promotion Co-op
Green Creek
Bear Creek
Former North Muskegon WWTP
Bear Lake
St. 148
St. 103
St. 104
Ryerson Ck.
Teledyne Cont. Motors
West Michigan Dock and Market Corp.
Shaw Walker
Mich. Foundry
Geerpress Wringer
Howmet #2
Port City Paints
Anaconda Wire and Cable
Breneman Inc.
Westran Corp.
S.D. Warren
Muskegon Piston Ring
St. 106
Muskegon Lake
St. 107
St. 105
Muskegon
Sealed Power
Lake Michigan
Clarke Gravely
Ruddiman Ck.
Keene Corp.
Enterprise Brass
Standard Oil Car Wash
Campbell, Wyatt and Cannon
Cannon Muskegon Corp
Great Lakes Plating
Industries
o = in
• = out
Communities
= in system
= out, but to enter within 20 years
= out, no plan
Michigan Spring
Roosevelt Park
Muskegon Heights
0 1 2
kilometers

Figure 59. Location and diversion status of point sources and municipalities in the Muskegon Lake basin.

Information regarding the major industrial and municipal discharges to Muskegon Lake is summarized in Table 20. Estimates of the nutrient loads were obtained from operating reports and Michigan Water Resources Commission industrial and municipal waste surveys. The four most significant pre-diversion discharges were the S.D. Warren Paper Company, the Storey (Ott) Chemical Company, and the Muskegon and North Muskegon Wastewater Treatment Plants. These discharges were diverted to the spray-irrigation system on the dates shown in Table 1.

Nutrient loads in the tributaries were calculated by multiplying the monitored river concentrations by the specific daily flow as calculated from the USGS measurements on the Muskegon River and Bear Lake Channel. The USGS flows were corrected to represent the total flow at a specific sample station. A multiplication factor was used to compensate for the increase in flow below the USGS station. This was derived from miscellaneous flow measurements and drainage area considerations. Because no consistent or reliable information was available regarding the split in flow between the various branches of the Muskegon River, concentrations were averaged from the different branches and multiplied by the total flow to obtain the loads. A summary of the calculated tributary loads is given in Table 21. Plots of the calculated Muskegon River loads are given in Figures 60 through 63. The emphasis of the load calculations was on total phosphorus and dissolved inorganic nitrogen because they were believed to be most important in evaluating the effectiveness of the diversion program. Emphasis has usually been placed on yearly averages, for reasons previously outlined in discussion related to White Lake. All conclusions derived from the yearly averages were also supported by seasonal analysis of the data.

The tributary nutrient loads from Bear Lake Channel were compared with those of the Muskegon River. This comparison demonstrated the insignificance of the load from Bear Lake Channel. The Bear Lake Channel load was typically 1% or less of the total phosphorus tributary load and less than 2% of the chloride load (approximately 3 kg P/day and 2500 kg Cl/day). No further consideration of its influence was pursued.

Patterns were observed in the seasonal variation of nutrient loads from the Muskegon River. The seasonal variation in nitrogen and phosphorus loads was consistent with the variation in concentrations. The chloride and silicon loads were highest in early spring and were associated with high river flows.

The primary focus in this investigation was to examine point and non-point nutrient sources to Muskegon Lake and evaluate the effects of the wastewater diversion on water quality. The sources of nutrients to Muskegon Lake and the potential and actual effectiveness of the sewage spray-irrigation system were also evaluated. To this end several additional computations were useful. Table 22 is an annual breakdown of point and non-point nutrient loads to Muskegon Lake. The point discharges

Table 20. ESTIMATED MUNICIPAL AND INDUSTRIAL LOADS TO MUSKEGON LAKE PRIOR TO DIVERSION IN KILOGRAMS PER DAY*

Source & Location of Discharge	Receiving Water	Chemical Oxygen Demand	Biochemical Oxygen Demand	Suspended Solids	Soluble Reactive Phosphorus as P	Total Phosphorus as P	Dissolved Inorganic Nitrogen as N	Total Nitrogen as N	Date of Diversion	Source
Campbell, Wyant, & Cannon Company, Muskegon	Ruddiman Creek	121	28.1	158	0.62	1.99	8.39	>8.39	—	MWRC 1973
Continental Motors, Muskegon	Muskegon Lake	418	134	391	1.16	2.06	2.7	14.4	—	MWRC 1967-69
Misco Division of Howmett Corporation, Muskegon	Ruddiman Creek	4.2	—	1.27	0.01	0.04	0.71	>0.71	—	MWRC 1973
Kaydon Bearing Keene Corporation, Norton Shores	Ruddiman Creek	4.0	—	0.6	0.01	0.07	0.32	>0.32	—	MWRC 1973
Michigan Foundry Supply Company, Muskegon	Muskegon Lake	8.4	—	18.6	0.0	0.0	0.41	>0.41	—	MWRC 1973
Muskegon Piston Ring Company, Muskegon	Ruddiman Creek	3.7	0.5	1.6	0.01	0.01	0.03	0.09	—	MWRC 1973
Naph Sol Refinery, North Muskegon	North Branch Muskegon River	561	72.6	—	1.86	2.88	0.0	13.4	—	MWRC 1967
Ott (Storey) Chemical Company, Muskegon	Middle Branch Muskegon River	3387	1328	95	0.13	0.81	425	>425	4/18/74	MWRC 1969
S.D. Warren Paper Company, Muskegon	Muskegon Lake	—	5625	5041	—	37.4	113.5	>113.5	6/4/73	MWRC 1966-67
Sealed Power Corporation, Muskegon Heights	Ruddiman Creek	34	—	358	5.4	6.5	10.4	>10.4	—	MWRC 1973
Westran Corporation, Muskegon	Muskegon Lake	29.7	—	19.5	—	0.08	2.3	>2.3	—	MWRC 1973
Muskegon Wastewater Treatment Plant	Middle Branch Muskegon River	—	—	—	—	93.2	562	1278	5/10/73	Muskegon Co. 1970 & 1975
North Muskegon** Wastewater Treatment Plant	Muskegon Lake	—	—	—	—	15.9	95.9	218	6/9/73	Muskegon Co. 1970 & 1975

*This table does not include industrial outfalls which prior to 1973 discharged to a municipal treatment system. It also does not include industrial outfalls discharging to private lagoons or septic systems. Nutrients loading to Muskegon Lake from these latter sources could not be quantified. Several other industries are known to discharge to surface waters in the area; however, no data on the quality of their discharges were available. Appendix F contains additional data.

**Load estimate based on Muskegon Wastewater Treatment Plant effluent quality and North Muskegon Wastewater Plant capacity.

Table 21. AVERAGE MUSKEGON LAKE TRIBUTARY LOADS IN KILOGRAMS PER DAY

Station	Year	Total Phosphorus as P	Nitrate + Ammonia as N
Muskegon River (at mouth)	1972*	273	1744
	1973	421	1268
	1974	274	1403
	1975	389	2306
Muskegon River (Station 102)	1972	164	686
	1973	189	717
	1974	314	1250
	1975	444	2189
Muskegon River (Station 143)	1973	166	723
	1974	188	1278
	1975	258	1970

---

*Estimated based on upstream water quality and point loads.

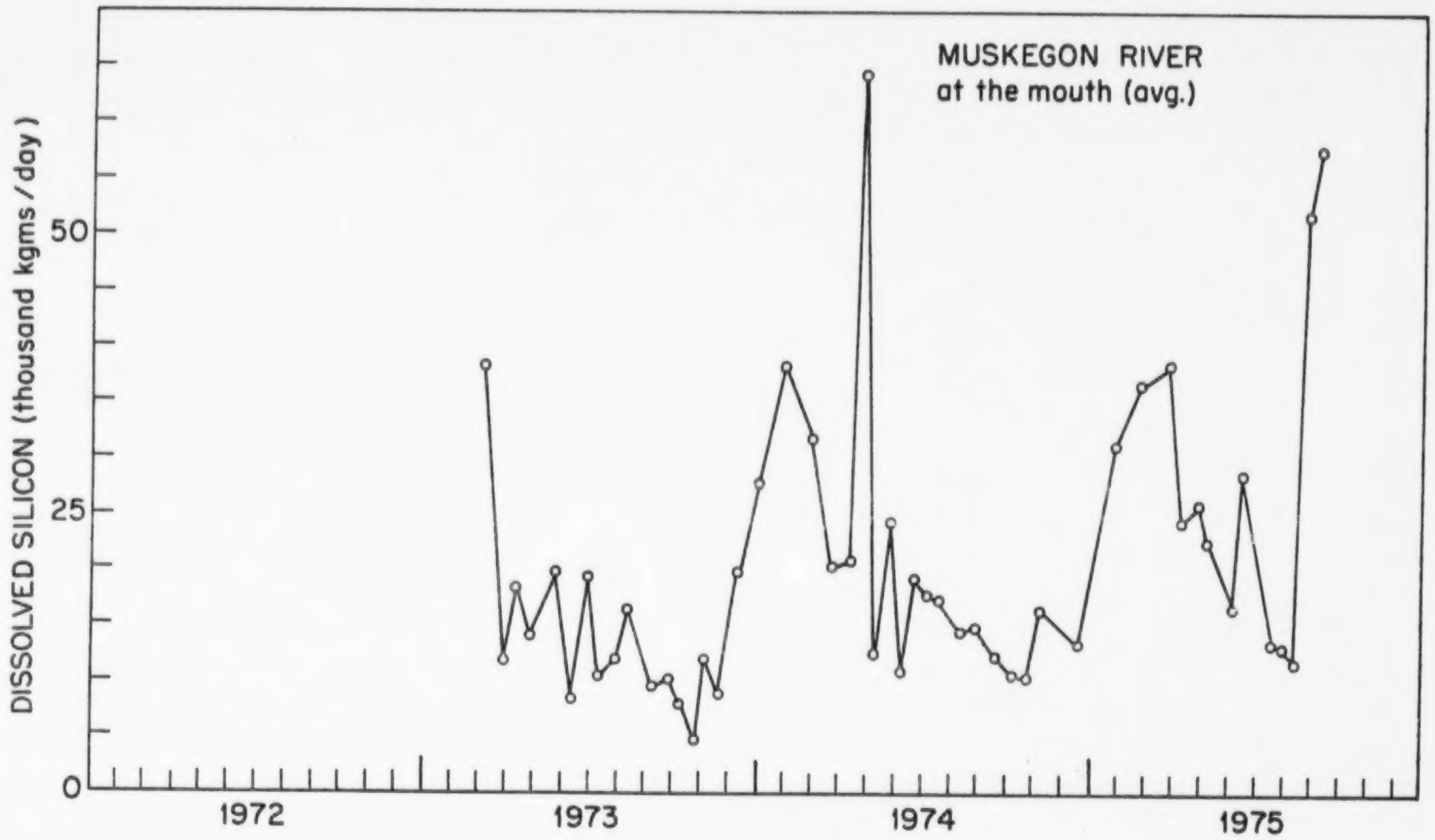


Figure 60. Silicon load of the Muskegon River, 1973-1975.

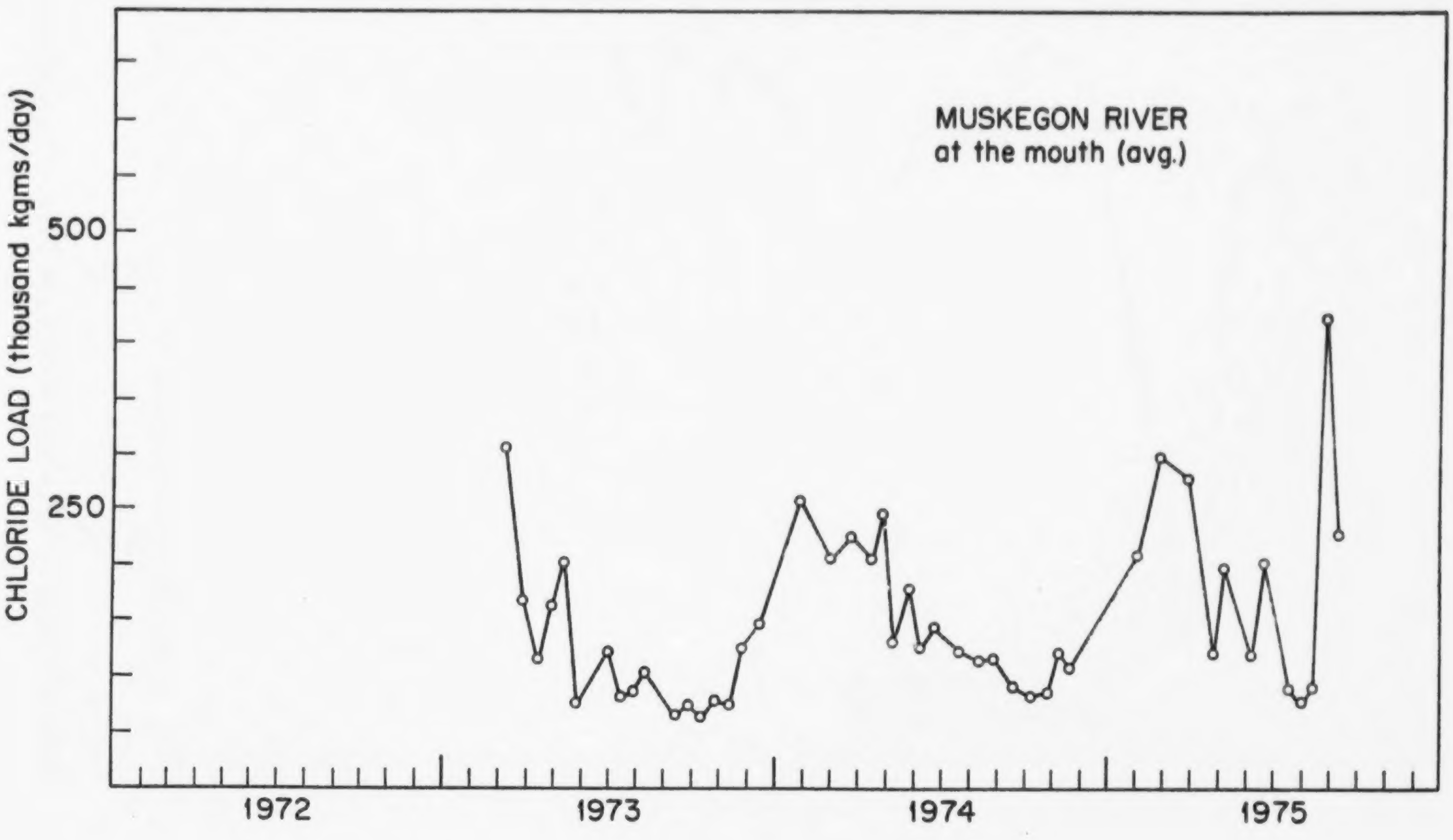


Figure 61. Chloride load of the Muskegon River, 1973-1975.

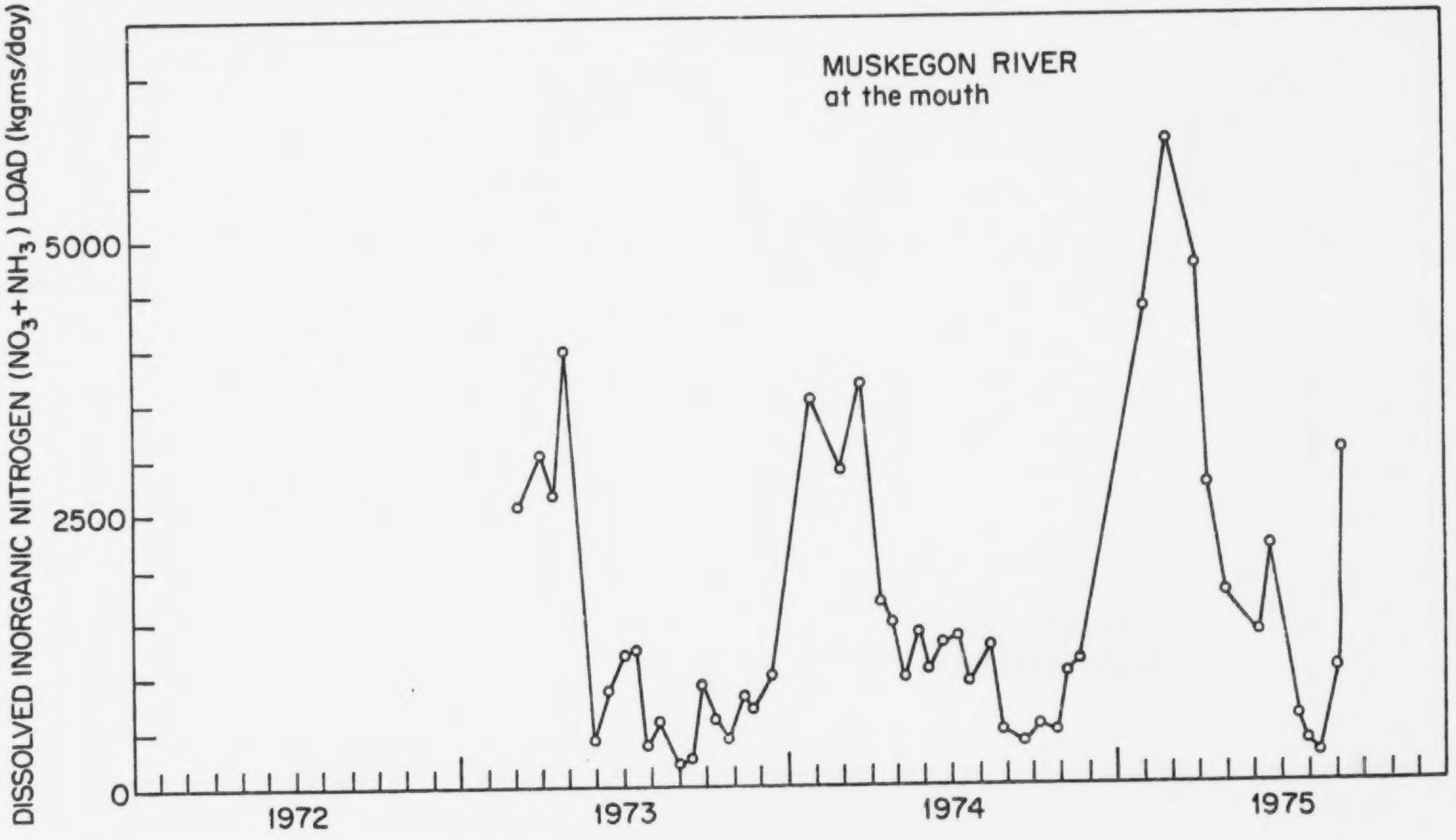


Figure 62. Dissolved inorganic nitrogen load of the Muskegon River, 1973-1975.

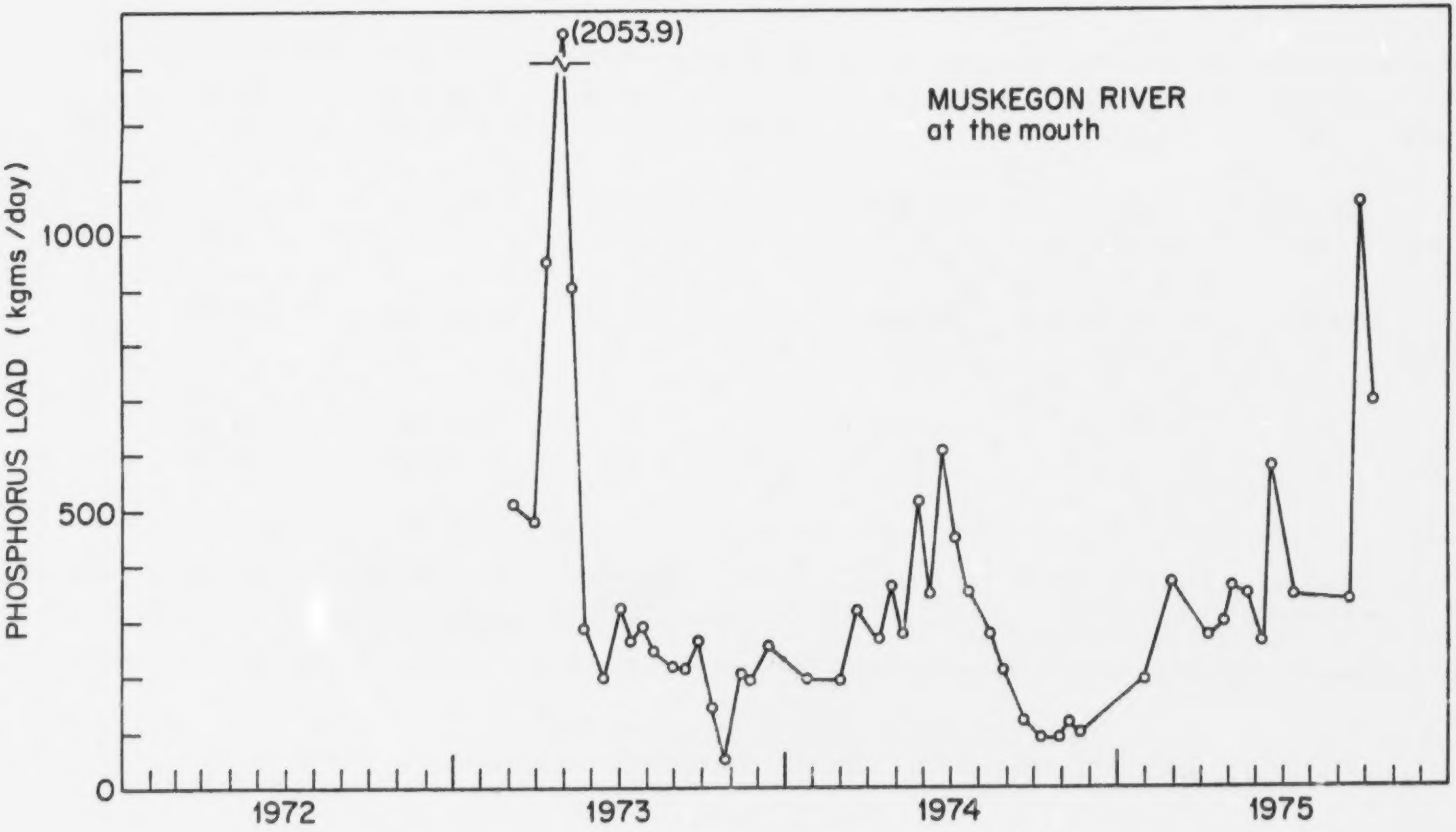


Figure 63. Total phosphorus load of the Muskegon River, 1973-1975.

Table 22. ESTIMATED TOTAL YEARLY NUTRIENT LOADS TO MUSKEGON LAKE, 1972-1975 IN THOUSAND KILOGRAMS PER YEAR

	Total Phosphorus				Dissolved Inorganic Nitrogen			
Year	Municipal & Industrial	Non-Point Muskegon River	Spray-Site Drainage	Total	Municipal & Industrial	Non-Point Muskegon River	Spray-Site Drainage	Total
1972	58.7 (47.1%)	65.8 (52.9%)	0	124.5	445 (61.8%)	275 (38.2%)	0	720
1973	27.9 (26.9%)	75.3 (72.6%)	.5 (.5%)	103.7	264 (47.8%)	282 (51.1%)	6 (1.1%)	552
1974	5.1 (4.8%)	98.95 (93.7%)	1.55 (1.5%)	105.6	45 (8.1%)	454 (81.4%)	59 (10.5%)	558
1975	5.0 (3.4%)	140.8 (95.5%)	1.55 (1.1%)	147.4	9 (1.1%)	747 (87.8%)	94 (11.1%)	850

were estimated from Table 19 and adjusted to reflect their actual duration of discharge. Spray-site loads were estimated from operating data presented in Appendix E. The non-point sources of nutrient loads from the Muskegon River were estimated using Muskegon River data which was not influenced by point industrial or municipal sources. The loads represent unidentified upstream loads which the authors chose to characterize as non-point. The calculations of the total loads to Muskegon Lake in this table did not include minor tributary drainage, urban runoff, or atmospheric inputs. A 1972-73 investigation by the USEPA (1975c) estimated that these nutrient sources amounted to only 1 or 2% of the total nitrogen and phosphorus load to Muskegon Lake.

The contribution of atmospheric inputs can be estimated from data collected by Johnson (1976). Using an average atmospheric phosphorus input of 20 kg P/km^2/yr, the total yearly contribution from the atmosphere is calculated to be 336 kg P/yr. This is an insignificant amount. The source of nutrients originating from storm runoff have been roughly estimated (without supporting data) by the Western Michigan Shoreline Federal 208 Planning Agency. This discharge into Muskegon Lake was through two primary drains, Four Mile Creek and Ruddiman Creek. This preliminary data suggest that the urban storm-runoff phosphorus load could be as much as 9% (9200 kg P/yr) of the total system load (Weaver, 1976). This is not in agreement with the USEPA estimate. Direct measurements are needed to quantify this source.

A second calculation performed on the data was intended to normalize the river loads to average flow conditions. The loads were normalized by multiplying each yearly average load by the ratio of the 52-year average annual flow to the specific annual average flow. This technique will normalize the effects of changing flow, but it does not normalize the effect of the variable intensity of the non-point source loads associated with changes in flow. The results of the normalized flow calculations are given in Table 23.

The potential effectiveness of the sewage diversion can be evaluated based on the four years of data collected on nutrient loads to Muskegon Lake. If 100% of the measured point discharge loads were eliminated, a 38% reduction in total phosphorus and 50% reduction in dissolved inorganic nitrogen is expected. These estimates were derived by averaging the non-point Muskegon River loads for 1972 to 1975 (Table 22 and comparing this to the industrial and municipal pre-diversion point discharges. (This does not include the unquantified storm water source or spray site drainage). If the normalized tributary loading data are used in this calculation instead of the observed loads, the percentage reductions calculated are 39% for total phosphorus and 52% for dissolved inorganic nitrogen. The nutrient load reduction expected in the Muskegon River load alone was 27% for phosphorus and 47% for dissolved inorganic nitrogen. These estimates were calculated using the normalized tributary loads. A comparison of the above calculation with data collected by the USEPA

Table 23. NORMALIZED YEARLY AVERAGE MUSKEGON RIVER LOADS IN KILOGRAMS PER DAY

Station	Year	Total Phosphorus as P	Nitrate + Ammonia as N
Muskegon River (at mouth)	1972	290	1970
	1973	340	1030
	1974	210	1090
	1975	320	1860
Muskegon River (Station 102)	1972	180	770
	1973	150	580
	1974	240	970
	1975	360	1790
Muskegon River (Station 143	1973	130	590
	1974	140	990
	1975	210	1610

(1975c) was not possible because the USEPA incorrectly surveyed and identified the municipal discharges and had no information on industrial discharges. It should be emphasized that a large share of the phosphorus reduction resulting from the Muskegon Wastewater Treatment Plant diversion was soluble reactive phosphorus. This form is considered to be immediately available for algal uptake.

Silicon and chloride loads were observed to increase between 1973 and 1975. The silicon loading from the Muskegon River was 14,200 kg/day in 1973; 20,600 kg/day in 1974; and 28,600 kg/day in 1975. The chloride load was 110,000 kg/day in 1973; 150,000 kg/day in 1974; and 197,000 kg/day in 1975. These increases are primarily a reflection of the increased flow.

The actual measured yearly average nutrient loads were compared to evaluate the effectiveness of the diversion spray-irrigation project. Examining the non-normalized load data in Table 22, a decrease in nutrient (total phosphorus and inorganic nitrogen) loads is seen in 1973 and 1974 but an increase is observed in 1975. This trend is also observed if normalized load data are used. The approximate 20 to 25% reduction is probably a reflection of the sewage discharge diversion. The percentage reduction was not the anticipated 39% phosphorus and 50% dissolved inorganic nitrogen reduction. Differences in flow and in non-point loads cause this deviation. A low flow year occurred in 1972 and thus the pre-diversion calculated loads are likely to be lower than the average. The increase in load observed during 1975 is coincidental to the nutrient spray-irrigation project. An evaluation of the calculated loads at the two upstream stations on the Muskegon River demonstrate this. This is further supported by previous analysis of river concentration data. The increase in the 1975 load is a consequence of increased flow and increased concentration upstream of the discharge site.

The nutrient load calculated at Station 143 represents nutrients derived from unidentified upstream sources. The differences between the loads at Stations 143 and 102 should reflect the effect of the spray-irrigation drainage. From these data it is seen that the increase in loads observed in 1975 at the river mouth can be accounted for by an increase in load from upstream sources. No dramatic change in load was observed below Station 143 except for the influences of former point discharges. A small portion of the 1975 increase in load could be a consequence of the availability of loading data for 1975, which did not include the fall period. The 1975 loading increase is also observed in a seasonal analysis of the data.

No conclusive effects on receiving water quality were observed to result from the spray-irrigation drainage. Dissolved inorganic nitrogen loads did not increase downstream of the Mosquito Creek discharge. Total phosphorus loads were noted to increase but this was not definitely related to the diversion project. The drainage from the irrigation site was measured by Muskegon County and found to be relatively low in phosphorus with a concentration similar to that of the upstream Muskegon River waters. The increase in load from the high nitrogen waters of Mosquito Creek was not dramatic because of the low flow contribution. Calculations of yearly and monthly nutrient loads from the spray site drainage is available in Appendix E. These loads are compared to the other point and non-point loads in Table 22. The spray drainage phosphorus load was at most 1.5% of the total lake load; the dissolved inorganic nitrogen load was at most 11%.

In all prior discussions average yearly loading data were used for analysis. The reasons for this approach have been previously described. If loading data were evaluated seasonally, the calculated percentage reduction from the sewage diversion project would vary slightly. Examining the loading data for the months June through September, the potential percentage reduction is calculated to range from approximately 30 to 45% depending on the year. Averages for the spring period show potential percentage reductions of the same order of magnitude. Similar calculations for dissolved inorganic nitrogen show potential reductions in the summer to be as much as 50 to 60% depending on the year and in the spring a much lesser percentage.

An additional evaluation of the effectiveness of the diversion project was made by comparing the flow weighted average tributary concentrations resulting from the calculated nutrient loads (see Table 24). These data indicate that a reduction in nutrient concentrations in Muskegon Lake is expected as a consequence of the diversion. The increase in calculated concentration for 1975 is not believed to be a consequence of the diversion spray-irrigation program.

An additional quantitative evaluation of the effect of the observed nutrient loads to Muskegon Lake can be obtained using the Vollenweider (1975) model. This model was briefly discussed in the section on nutrient loads to White Lake. A load greater than the model-defined dangerous rate represents one causing or maintaining eutrophic conditions while a loading rate below the permissible rate represents a condition suitable for maintaining oligotrophic conditions. A loading rate between the two would be considered consistent with a mesotrophic system. In Muskegon Lake the permissible rate is calculated to be 1.08 g P/m^2/year while the dangerous rate is 2.16 g P/m^2/year. The pre-diversion phosphorus load to Muskegon Lake is 3.7 times greater than the dangerous rate and 7.4 times greater than the permissible rate. According to this model the reduction in phosphorus load resulting from the diversion project is not sufficient to correct the eutrophication problem in Muskegon Lake.

Table 24. FLOW WEIGHTED AVERAGE CONCENTRATIONS OF TOTAL PHOSPHORUS AND DISSOLVED INORGANIC NITROGEN IN TRIBUTARIES TO MUSKEGON LAKE

Year	Total Phosphorus (μg P/ℓ)	Dissolved Inorganic Nitrogen (μg N/ℓ)
1972	65.8	382
1973	39.8	213
1974	38.2	204
1975	56.0	325

The rate at which nutrients are leaving the system can be calculated by multiplying the channel concentration by the flow from the lake. A summary of these flux calculation with comparisons to the nutrient loads to Muskegon Lake is given in Table 25. The calculated lake retention of phosphorus is low, particularly in contrast to White Lake and other eutrophic systems. This may be a response of the system to non-steady state conditions, intermittent mixing, or shipping traffic. It is possible that the effects of reduction in nutrient load can be affected by phenomena such as sediment nutrient release which tend to buffer the system. This may have been observed in 1973 when a reduction in phosphorus load to the lake occurred. The percent retention in 1973 was negative indicating a nutrient contribution from within the lake. The dissolved inorganic nitrogen retention was calculated to be less during the post-diversion period. This observation is impossible to evaluate without information on total nitrogen because of the effects of algal uptake and ammonification. No change in retention of dissolved silicon was observed. The high retention of silicon was a likely consequence of algal uptake of the dissolved silicon.

Summary -- The Muskegon River is the major hydrologic and tributary nutrient source to Muskegon Lake. The former municipal and industrial discharges were found to be significant contributors of nutrients to Muskegon Lake. Estimates of the potential nutrient load reduction resulting from the diversion program are 38% for phosphorus and 50% for dissolved inorganic nitrogen. Approximately 20 to 25% reduction in nutrient loads was actually observed following diversion. The difference is attributed to variation in river flow and non-point loads during the period of the investigation. Lower phosphorus, ammonia, and iron concentrations were observed immediately subsequent to diversion. The effects of changing non-point nutrient loads to Muskegon Lake were observed (particularly in 1975) to have the potential to partially or completely mask the nutrient reduction resulting from the diversion project. Calculations for Muskegon Lake using the Vollenweider model suggest no anticipated change from its present eutrophic status.

LAKE-RELATED CONSIDERATIONS

Spatial and Seasonal Distributions -- In a manner similar to that for White Lake, two stations in Muskegon Lake (Stations 103 and 106) were selected for the evaluation of horizontal variation in chemical characteristics. The total dissolved phosphorus and total inorganic nitrogen data were evaluated (see Figures 64 and 65). Station 103 is located in the extreme eastern portion of Muskegon Lake and is influenced by the Muskegon River, an important tributary. Station 106 is located in the western portion of the lake and should reflect overall conditions.

Table 25. A SUMMARY OF THE YEARLY AVERAGE FLUX OF NUTRIENTS LEAVING AND ENTERING MUSKEGON LAKE IN KILOGRAMS PER DAY

	Year	Total Phosphorus as P	Nitrate + Ammonia as N	Silicon as Si
Leaving	1972	260	1100	5400
	1973	360	700	6200
	1974	260	1200	10000
	1975	340	2100	18900
Entering	1972	340	1900	--
	1973	280	1500	14200
	1974	290	1500	20600
	1975	400	2300	28600
Percent Retained in Muskegon Lake	1972	24%	43%	--
	1973	-28%	46%	49%
	1974	11%	20%	49%
	1975	15%	9%	34%

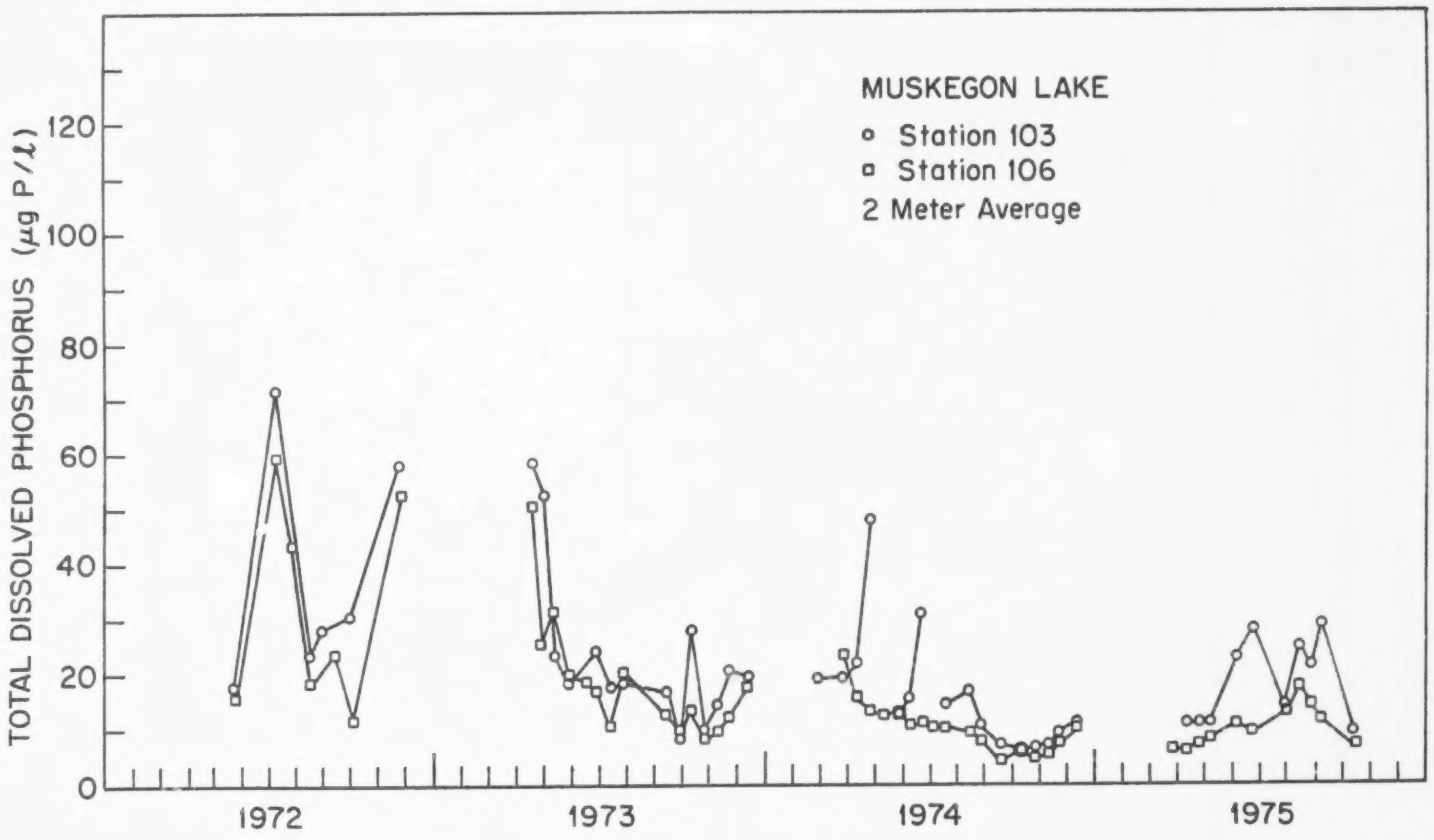


Figure 64. Total dissolved phosphorus concentrations at Stations 103 and 106 in Muskegon Lake, 1972-1975.

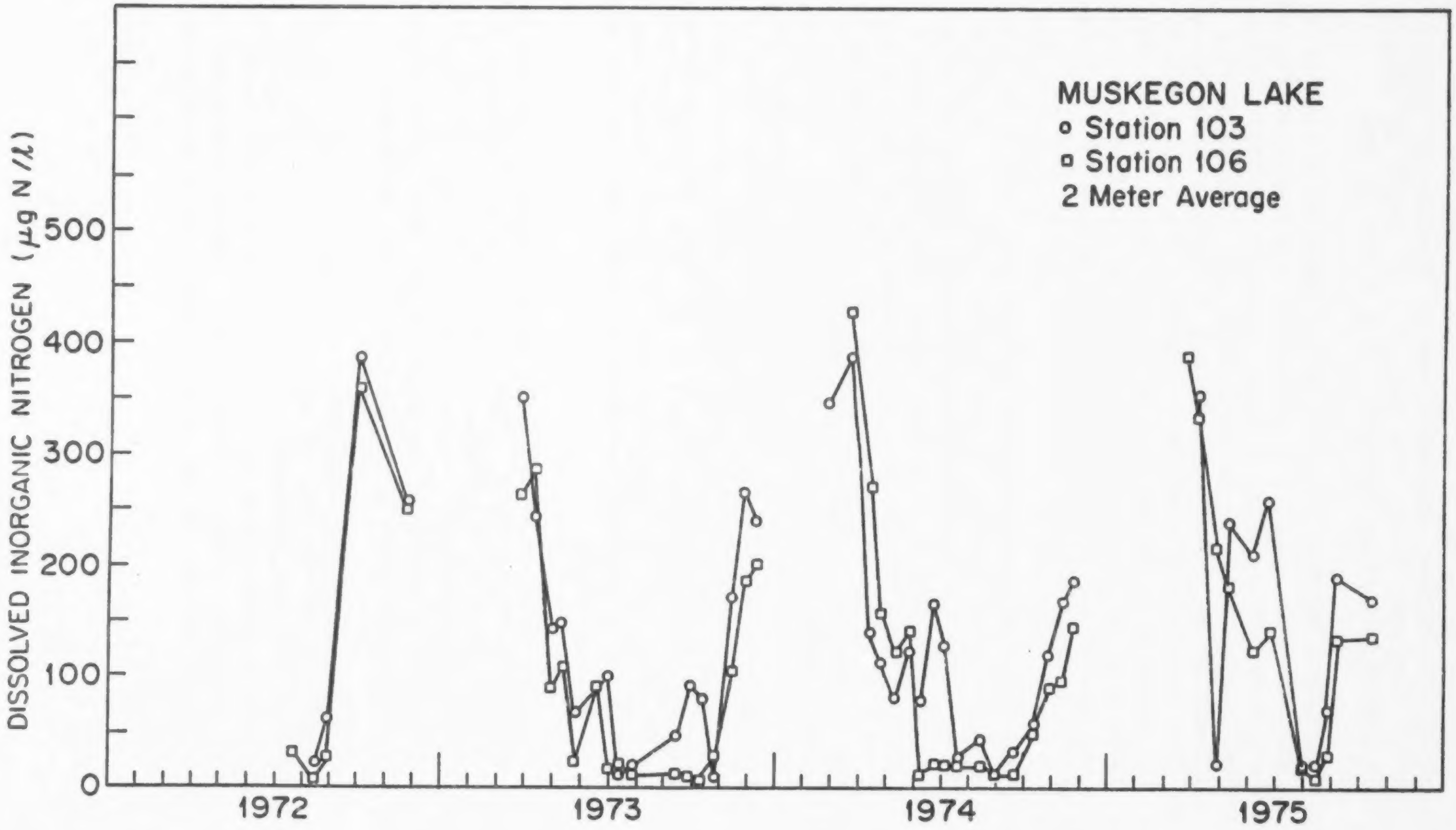


Figure 65. Dissolved inorganic nitrogen concentrations at Stations 103 and 106 in Muskegon Lake, 1972-1975.

An examination of total dissolved phosphorus data reveals that concentrations at Station 103 are generally higher than those at Station 106. Levels are usually 30 to 50% higher at Station 103, but twofold elevations were observed. The higher concentrations at Station 103 reflect the influence of the Muskegon River, the largest contributor of phosphorus to the lake and algae uptake between Stations 103 and 106.

Dissolved inorganic nitrogen concentrations were quite similar at Stations 103 and 106. The exception was during the summer months when Station 103 had concentrations of total inorganic nitrogen 100-150 μg N/ℓ greater than Station 106. It is postulated that this is also a result of the influence of the river. The Muskegon River contributes waters with very high levels of nitrate and ammonia. The excess dissolved inorganic nitrogen is utilized by algae with passage through the lake and has disappeared by the time it reaches Station 106. The difference is not as notable during the spring and fall because of the large influx of inorganic nitrogen from the bottom waters due to mixing.

Surface water (2m) concentrations of nitrate ($NO_2^- + NO_3^-$) in Muskegon Lake are a function of loading, algal uptake, and replenishment from the bottom waters. Maximum concentrations (200-400 μg N/ℓ) of nitrate occur in the winter (December through March) during high load and low uptake periods. These levels are rapidly reduced during the spring (March to June) (see Figure 66). Surface water nitrate concentrations approach zero during the summer due to algal uptake. Peak surface water ammonia concentrations occur during the winter months when phytoplankton uptake is low (see Figure 67). During the spring and summer months algal assimilation reduces ammonia concentrations to near the limit of detection. Short-term increases in summer surface water ammonia concentrations are probably due to the transfer of ammonia from the bottom waters during temporary periods of disruption of thermal stratification resulting from storms or ship traffic. At times higher surface water concentrations of ammonia were observed for Muskegon Lake than for the Muskegon River, its major tributary.

Nitrogen-fixation, thermocline disruption, and other processes are the likely cause of this phenomenon. The surface water inorganic nitrogen depletions observed in Muskegon Lake present the possibility that, like White Lake, the Muskegon Lake system is nitrogen limited. This is supported to an extent by the presence of nitrogen-fixing blue-green algae. A USEPA (1975c) bioassay study also confirmed this hypothesis.

Nitrate concentrations in the bottom waters follow a cycle of depletion and replenishment. The pattern is closely tied to surface water dynamics. Bottom water nitrate concentrations start from very high levels (200 - 400 μg N/ℓ) which are a result of elevated winter loads, nitrification and low plant uptake. Nitrate then rapidly decreases to near zero levels in the spring due to algal uptake. The spring phytoplankton

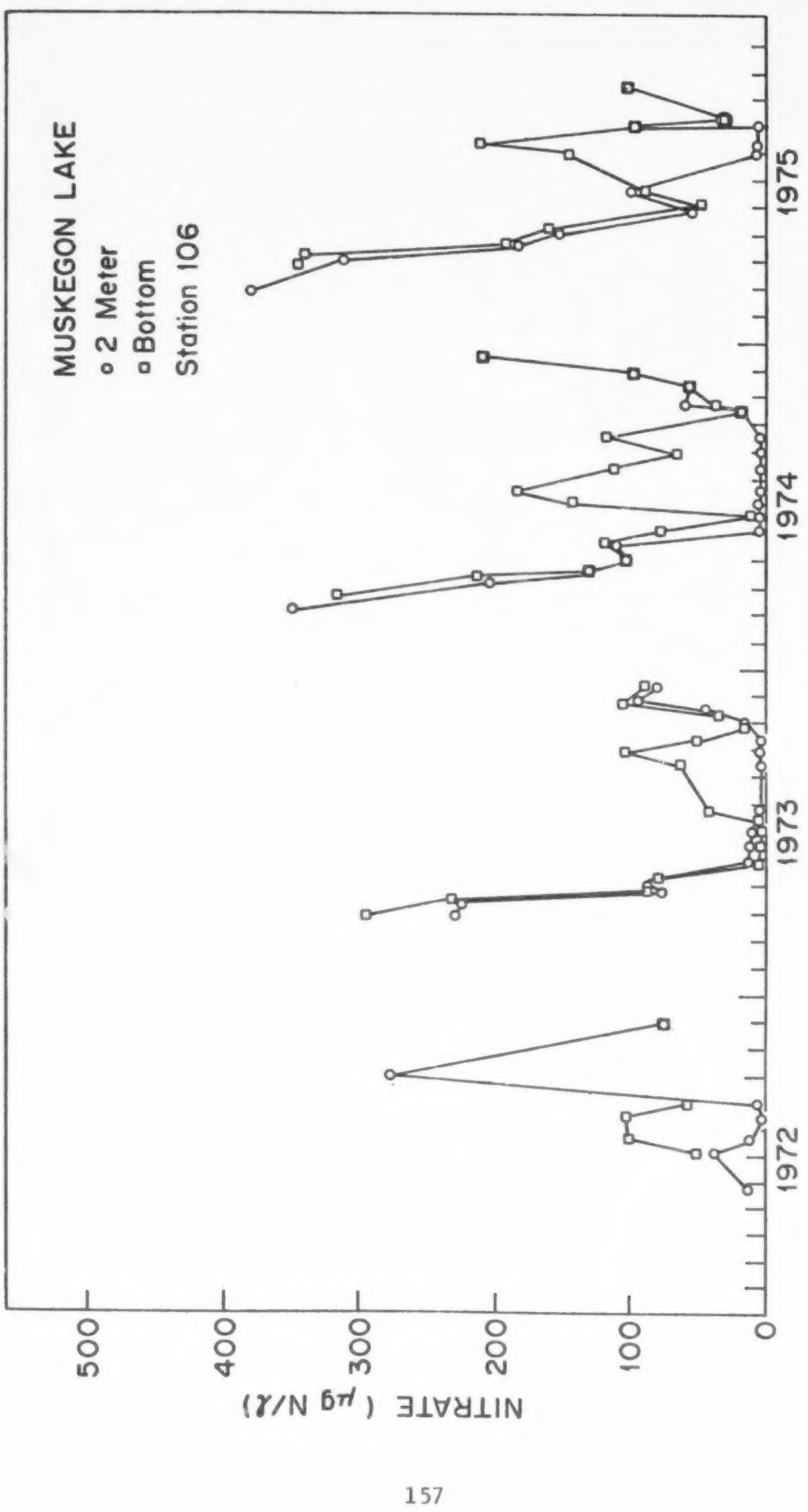


Figure 66. Nitrate concentrations in Muskegon Lake, 1972-1975.

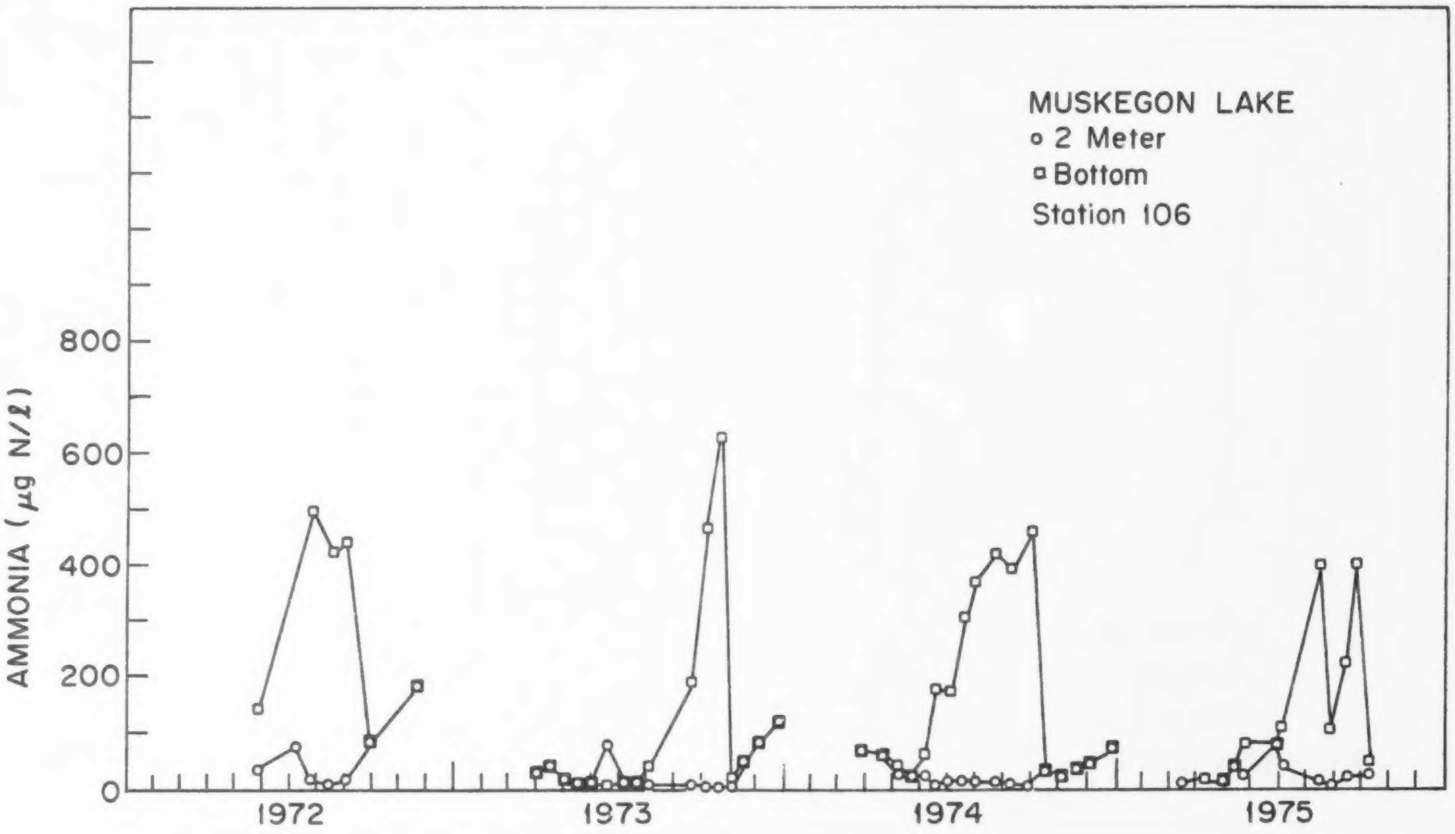


Figure 67. Ammonia concentrations in Muskegon Lake, 1972-1975.

bloom has an effect on the bottom waters only until such time as stratification occurs. With the onset of stratification bottom waters are no longer effected by uptake. Nitrate concentrations are occasionally replenished by nitrification. Dissolved oxygen was often available in the hypolimnion at depths greater than 1 meter from the bottom. Waters significantly above the lake bottom are affected less by sediment oxygen demand and more by sporadic mixing with the epilimnion. Temporary but frequent disruptions of the thermocline were observed resulting from ship traffic, storms, or other environmental phenomena (See Figure 68). The introduction of oxygen containing epilimnetic waters and intrusion of Lake Michigan waters (high in dissolved oxygen and nitrate) were sufficient to support nitrate concentrations in the bottom waters, permit some nitrification and retard or prevent significant denitrification. Later in the period of stratification the oxygen content of the hypolimnion is further reduced and some denitrification may occur. The result is a short-lived decrease in bottom water nitrate concentrations. During and after turnover nitrate is replenished in both surface and bottom waters through nitrification of hypolimnetic ammonia.

Ammonia concentrations in the bottom waters increase as a result of organic decomposition (largely during the anoxic summer months) and decrease with nitrification, and the mixing and assimilation period. Very high levels of ammonia (400 - 700 μg N/ℓ) are generated in the bottom waters of Muskegon Lake during the summer months. Nitrogen flux (transfer between forms) in Muskegon Lake approaches 1 mg N/ℓ in a given season. The magnitude of the nitrogen dynamics (winter nitrate peaks, ammonia buildup in summer, assimilation, nitrification) is notably similar for Muskegon and White Lakes.

As was the case with White Lake, three forms of phosphorus were measured for Muskegon Lake: soluble reactive, total dissolved, and total phosphorus. Generally the soluble reactive phosphorus accounted for much less than half of the dissolved component and the total dissolved phosphorus represented approximately one-third of the total phosphorus in the lake. The remaining two-thirds, the particulate fraction, includes organisms and detritus.

Concentrations of soluble reactive phosphorus and total dissolved phosphorus in the surface waters at 2 meters followed similar seasonal patterns (see Figures 69 and 70). Levels of total dissolved phosphorus are highest (30 - 40 μg P/ℓ) during the winter months (December through March) and then decrease during the spring and summer to concentrations of 5 - 10 μg P/ℓ. The supply of total dissolved phosphorus in the surface water is never reduced below 5 - 10 μg P/ℓ, indicating that this nutrient may not become limiting in Muskegon Lake. Soluble reactive phosphorus concentrations, although following a similar pattern, are generally lower (below 7 μg P/ℓ). The levels of soluble reactive

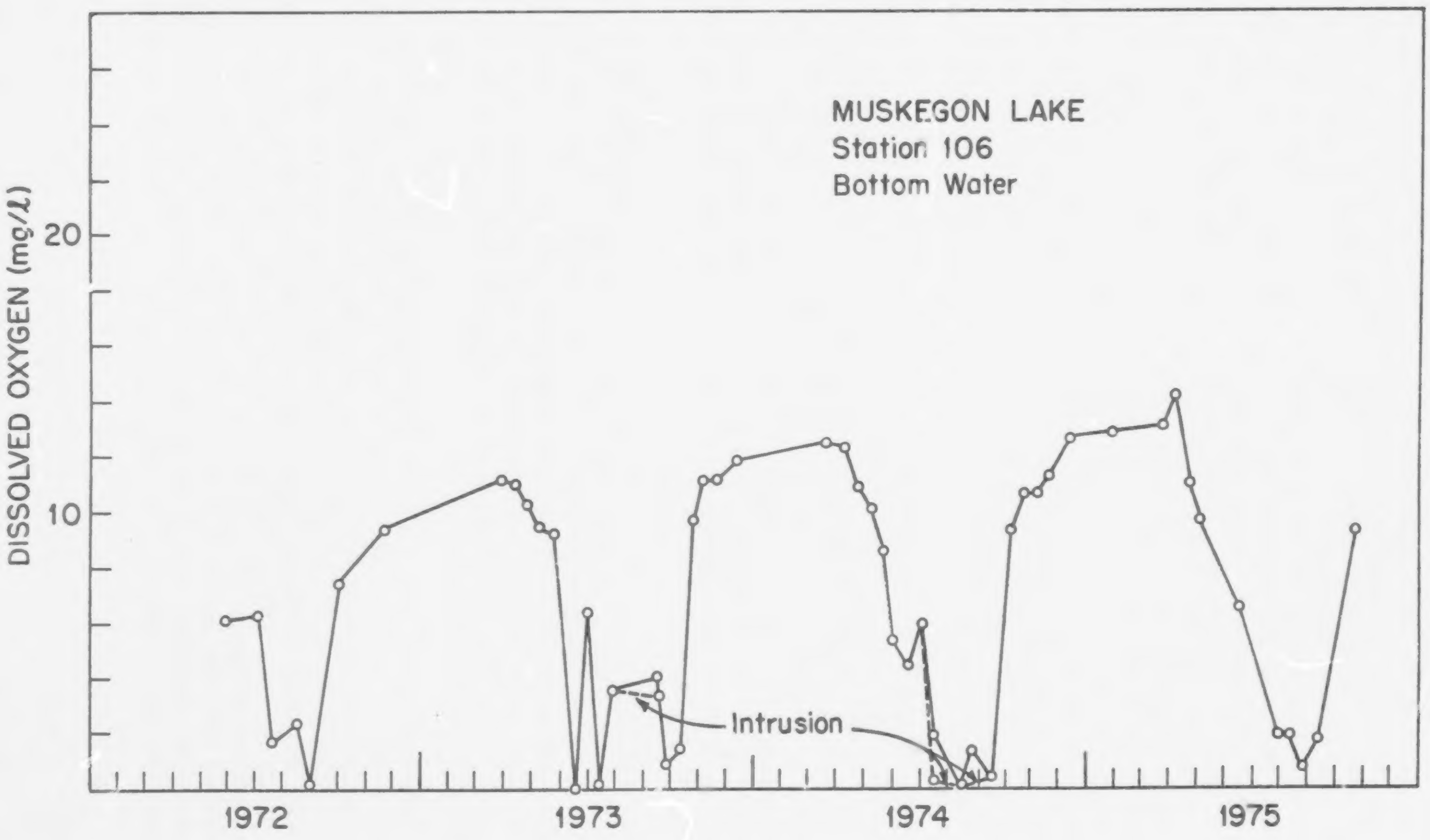


Figure 68. Dissolved oxygen in the bottom waters of Muskegon Lake, 1972-1975.

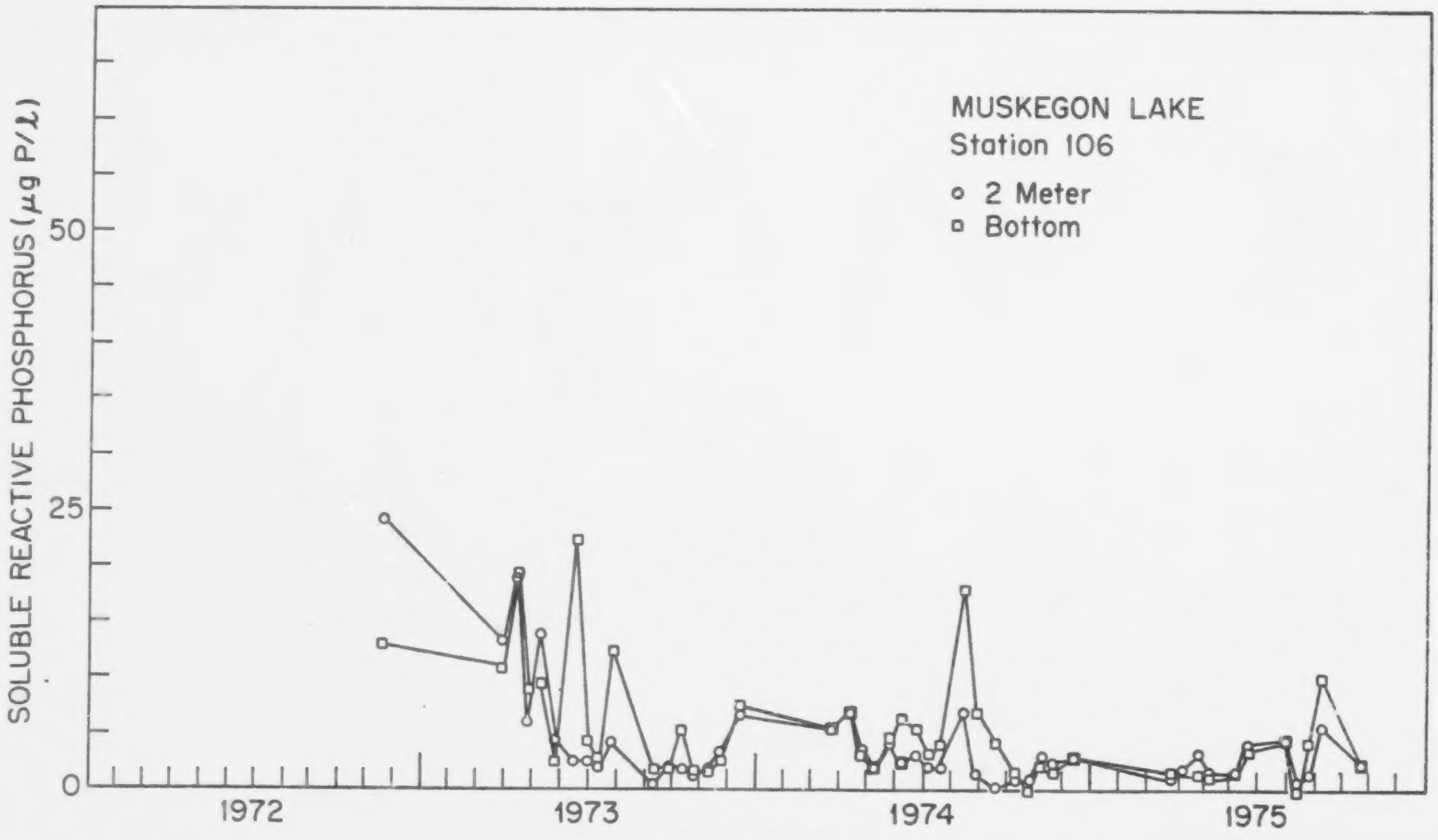


Figure 69. Soluble reactive phosphorus concentrations in Muskegon Lake, 1972-1975.

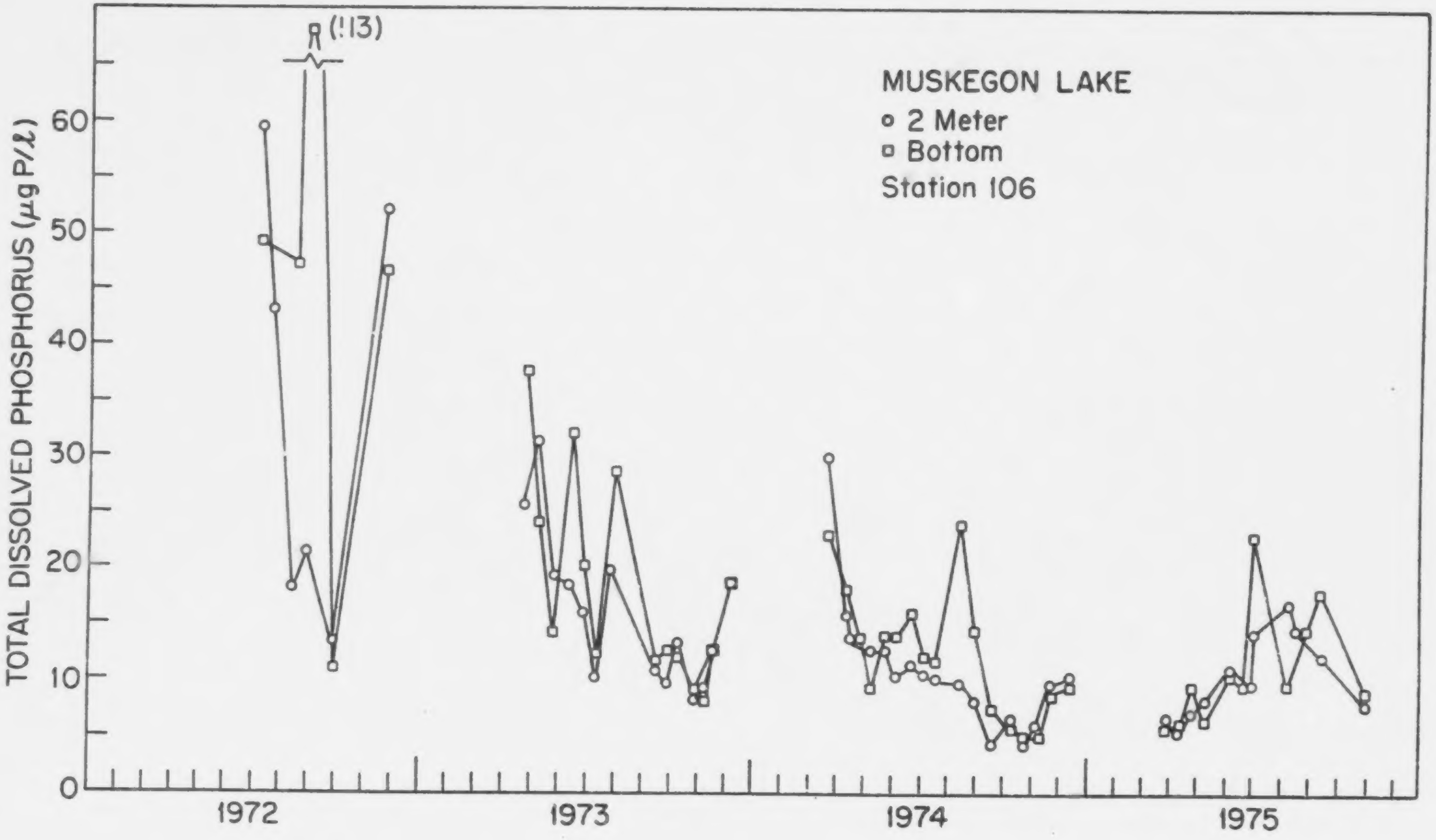


Figure 70. Total dissolved phosphorus concentrations in Muskegon Lake, 1972-1975.

phosphorus in surface waters are reduced to zero. The question of limitation by phosphorus then becomes one of the availability of the dissolved organic phosphorus fraction. This phenomenon is thought to be species specific (Keenan and Auer, 1974) and would require further investigation before any conclusions could be made.

Surface water total phosphorus data are somewhat incomplete. It is observed that surface water total phosphorus peaks occur at approximately the same time as peaks in phytoplankton abundance (see Figure 71). Concentrations through the remainder of the year are relatively stable. Since total phosphorus is the sum of the dissolved and particulate fractions, it should remain somewhat constant through the year while the relative contribution of the components change. Flux of phosphorus from the sediments to the biota and back to the sediments later in the year accounts for some of the seasonal variation in this parameter, along with variations in loads.

Bottom water concentrations of dissolved phosphorus in Muskegon Lake do not fluctuate as radically as those observed in White Lake. Summer peaks in dissolved (both total and soluble reactive) phosphorus were noted, but the frequency and particularly the magnitude of the peaks were reduced in comparison to White Lake. It is postulated that the reason for this is that the disruption of stratification by storms and the intrusion of well-oxygenated Lake Michigan water reduced both the duration and extent of oxygen depletion in the hypolimnetic waters; this results in the iron being maintained in the sparingly soluble ferric form. During the summers of 1973-1975 the release of dissolved iron to both the surface and bottom waters was noted (see Figure 72). This phenomenon was noted earlier in relation to nitrification processes in the bottom waters of Muskegon Lake. During the spring, late fall, and winter, bottom and surface dissolved phosphorus concentrations are quite similar. The fall peak in total phosphorus in 1973 is due to the particulate fraction; it is not clear why this peak is so much larger than peaks in other years.

Silicon concentrations in Muskegon Lake follow one of the most consistent seasonal patterns observed. Surface water values increase through the fall and winter months to a peak of 3 to 4 mg/ℓ in March. The spring diatom bloom then depletes this concentration through the spring and early summer to a minimum value of 0.5 to 1.0 mg/ℓ in June or July. Silicon levels remain low during the summer and early fall and then begin the replenishment period. Silicon barely reaches levels approaching limitation in Muskegon Lake (30 - 50 μg/ℓ) (see Figure 73).

Peaks in primary productivity did not follow chlorophyll *a* patterns in Muskegon Lake nearly as well as in White Lake (see Figures 74 and 75). Chlorophyll *a*, instead of producing a single (unimodal) peak, varied throughout the spring and summer of 1973-1975 in a more polymodal fashion.

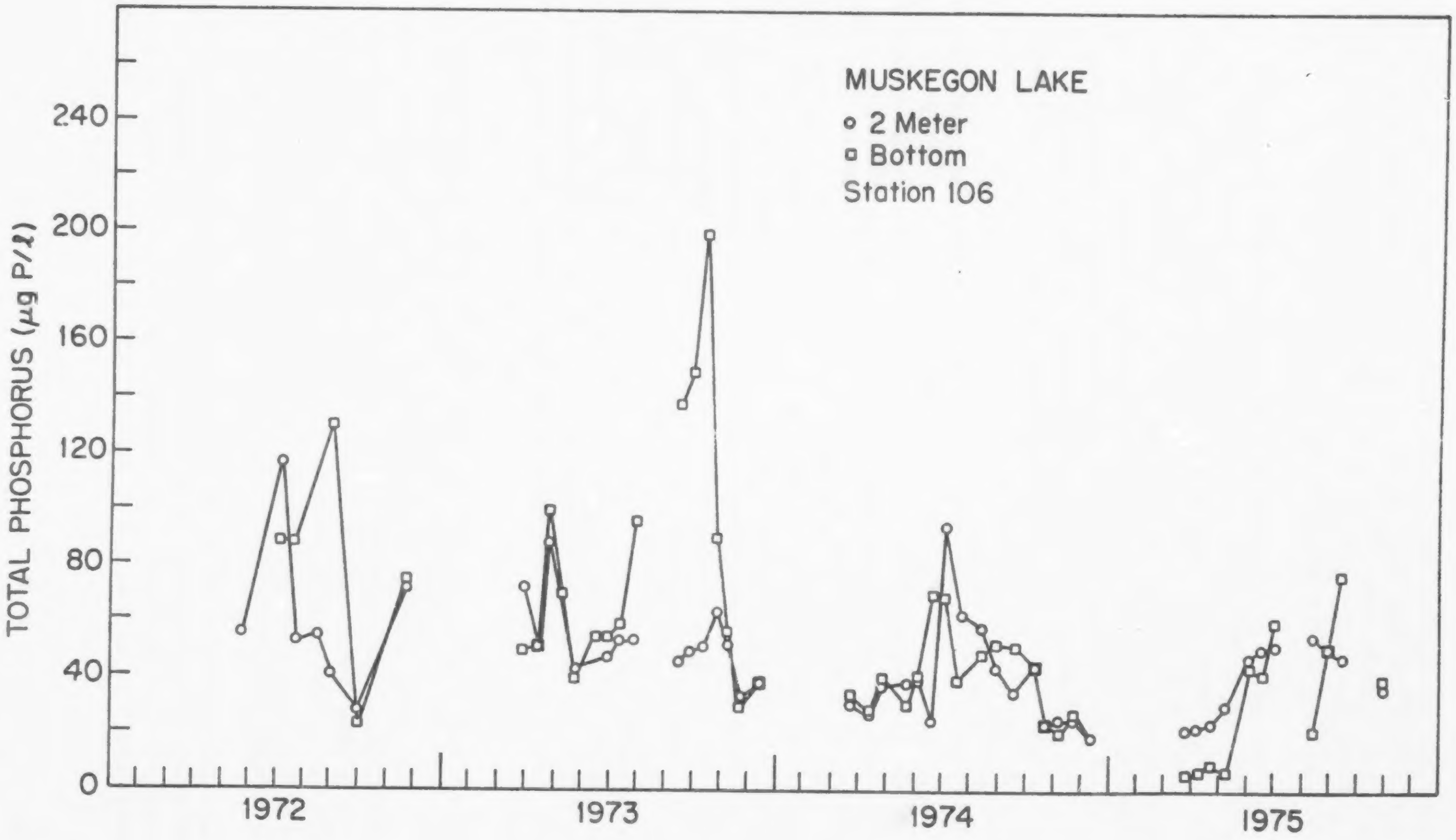


Figure 71. Total phosphorus concentrations in Muskegon Lake, 1972-1975.

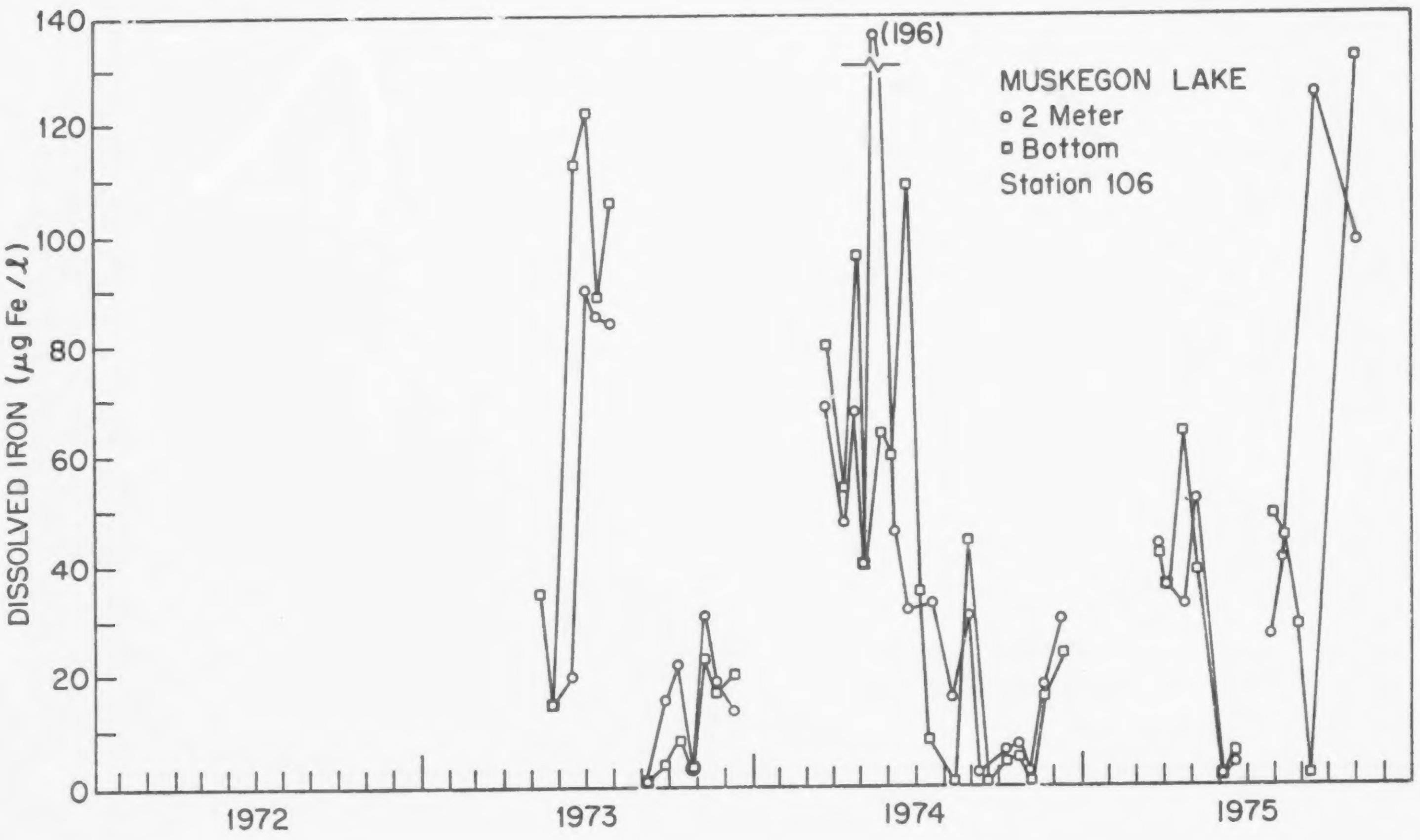


Figure 72. Dissolved iron concentrations in Muskegon Lake, 1972-1975.

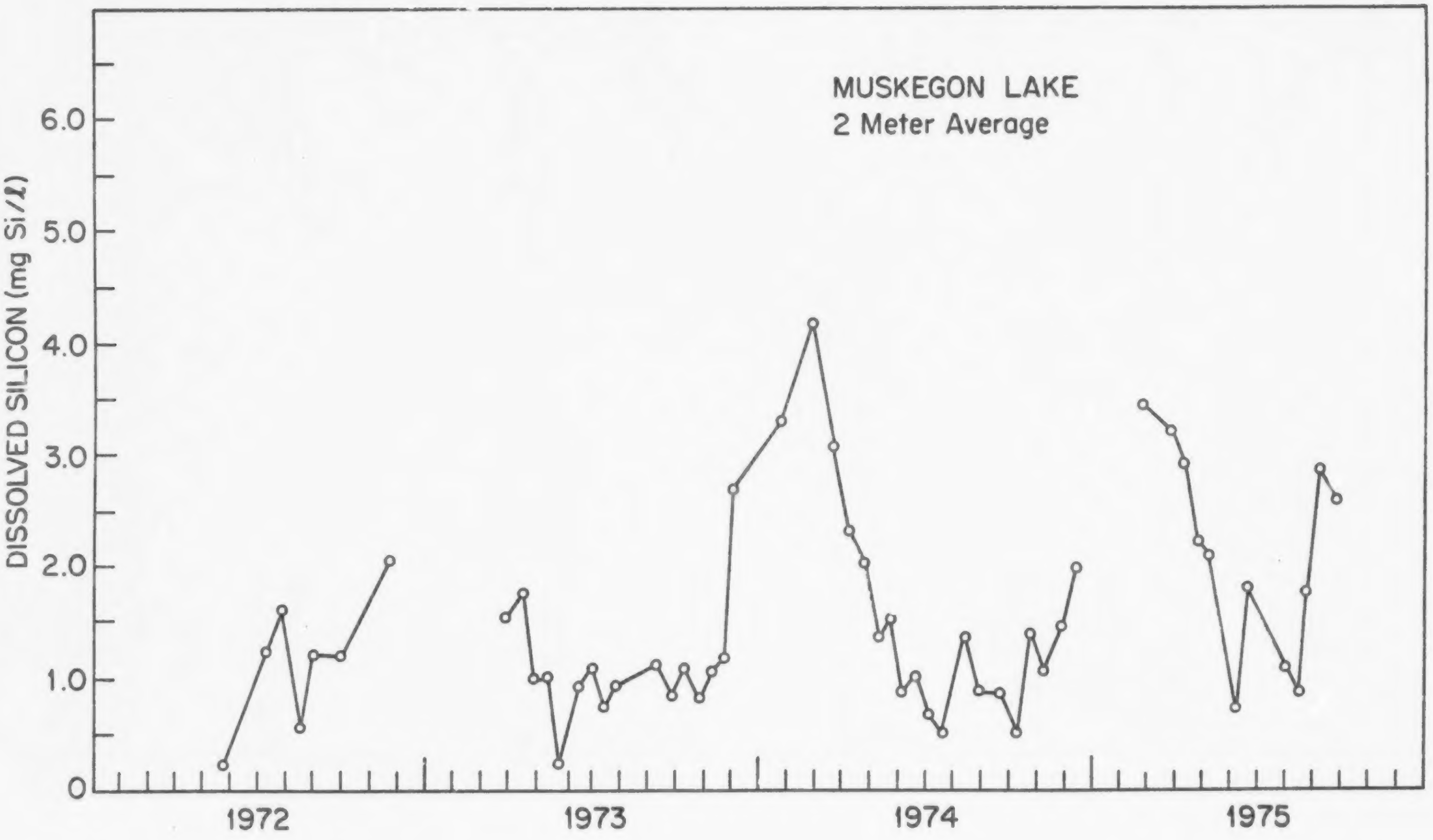


Figure 73. Dissolved silicon concentrations in Muskegon Lake, 1972-1975.

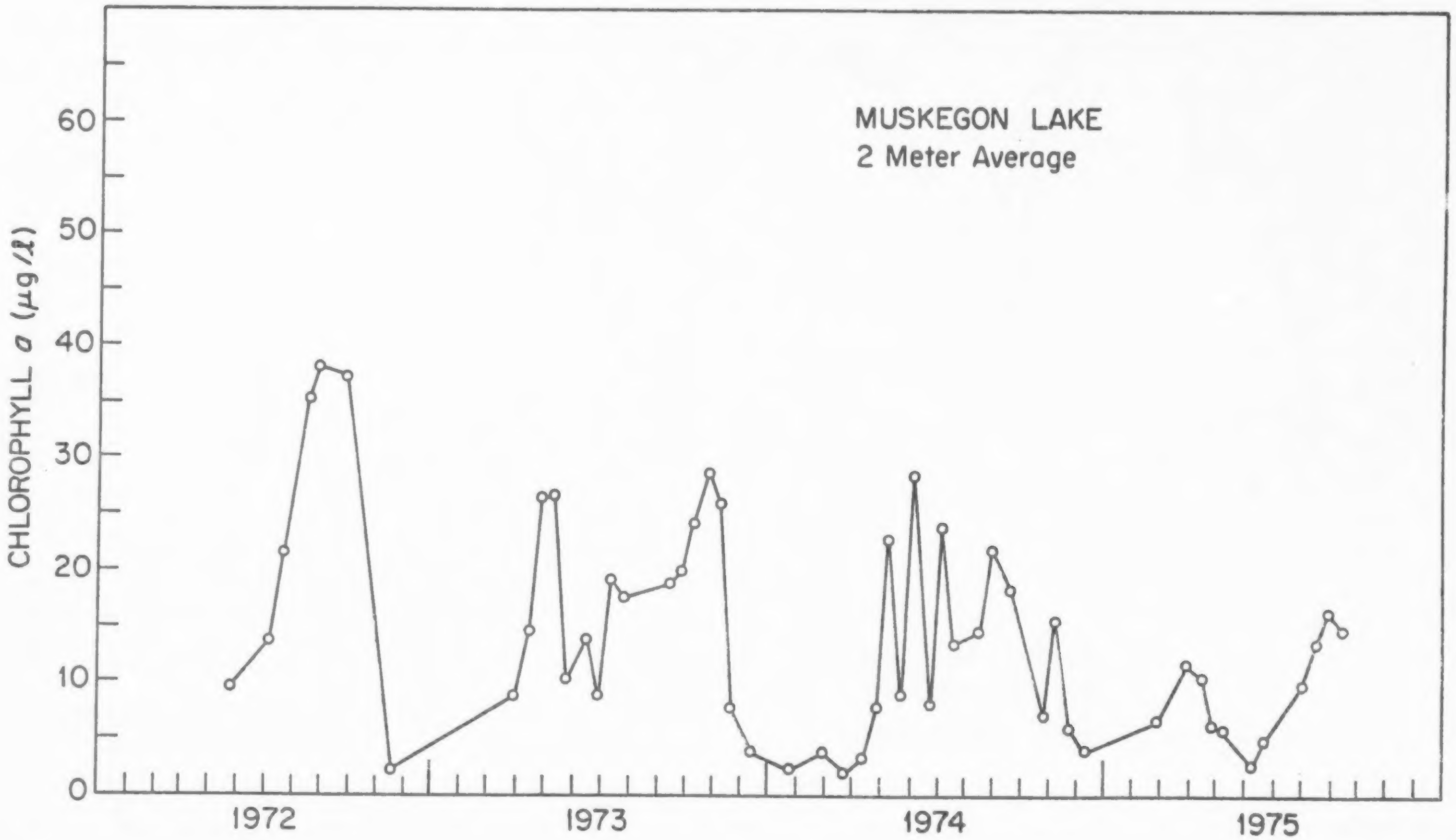


Figure 74. Chlorophyll *a* concentrations in Muskegon Lake, 1972-1975.

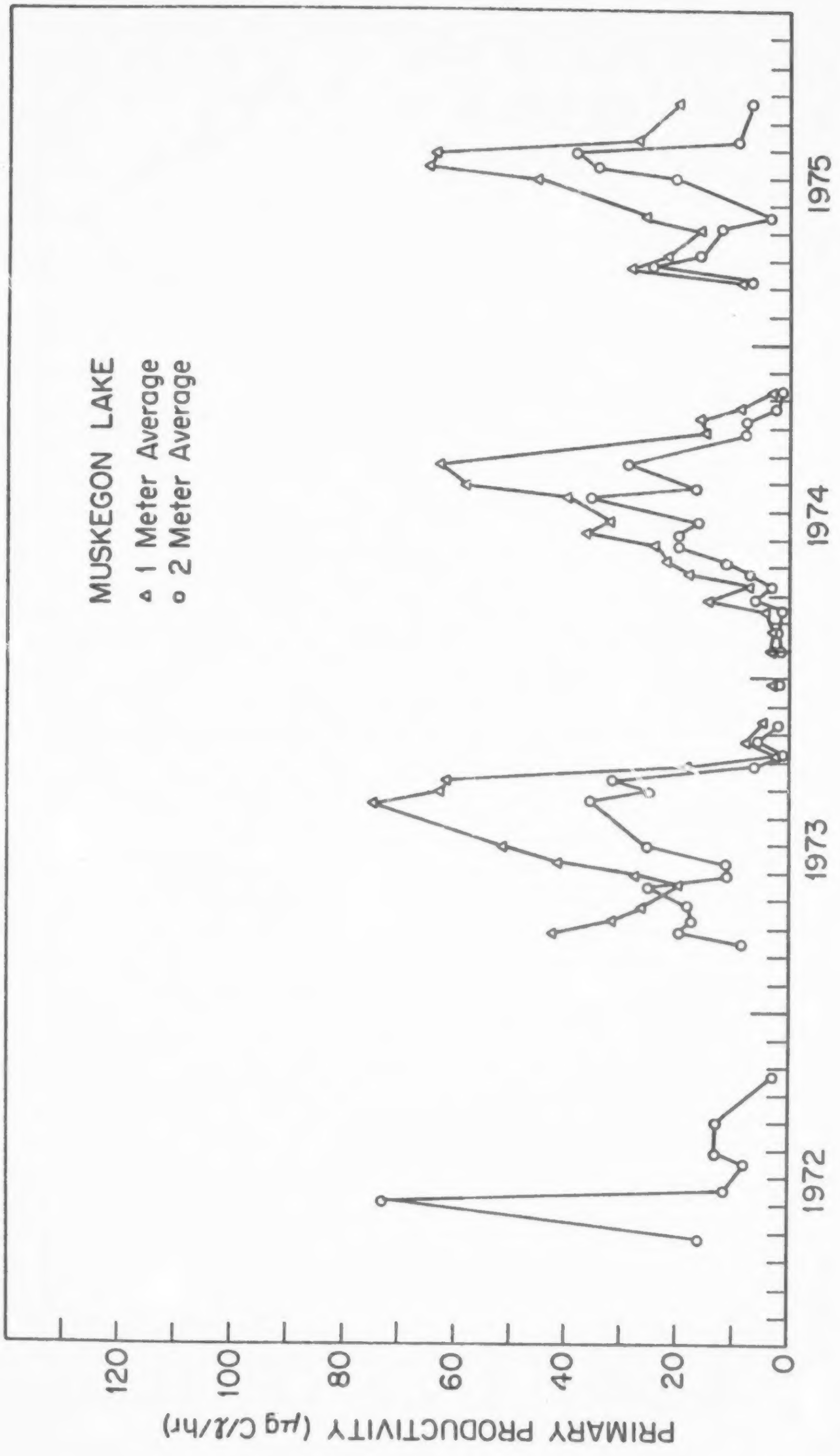


Figure 75. Primary productivity rates in Muskegon Lake, 1972-1975.

Data for 1972 showed only a single peak; however, sampling frequency may not have been adequate to identify other peaks. From an examination of chlorophyll *a* data alone, one might suggest the development of a bimodal (fall/spring), diatom dominated phytoplankton. The primary productivity values, however, display unimodal (late summer, early fall) peaks. The unimodal primary productivity patterns, at the peak, represent the point in time where the combination of standing stock, light, and nutrients is optimum. Obviously, it is not required that this match the chlorophyll *a* peaks.

Nutrient ratios of dissolved inorganic nitrogen to total dissolved phosphorus were calculated for Muskegon Lake as they were for White Lake. The winter N/P ratio in the surface waters was 13-14:1 in 1974 and 1975. The bottom water winter N/P ratios were 27-30:1 for this same period. These ratios indicate stoichiometric phosphorus limitation in the euphotic zone but are approaching nitrogen limitation in the surface waters. Nitrogen levels are generally higher than phosphorus levels in the winter since the iron-phosphorus dissolution mechanism is inoperative with oxygen present.

The summer surface N/P ratios in Muskegon Lake reach levels of 1:1 or less. This is due to the total depletion of nitrate and the reduction of ammonia to levels below the limit of detection. Muskegon Lake waters are then nitrogen limited in the summer months. The ratio of winter dissolved inorganic nitrogen to summer total dissolved phosphorus in the bottom waters was 19:1 in 1974 and 15:1 in 1975. Again these are values slightly indicating phosphorus limitation but approaching nitrogen limitation.

Another analysis of N/P ratios can be made by examining the system loads. Although this does not consider such factors as system recycle or retention, it is still a useful technique. Calculation of the N/P ratio based on average yearly loads result in ratios of 5 to 10:1. This indicates a well-balanced to nitrogen limited nutrient status. Analyses of the spring loading data suggest slightly higher ratios favoring stoichiometric nitrogen limitation. The summer nitrogen limitation which was suggested by summer loading N/P ratios and lake surface water N/P ratios is further supported by USEPA (1975c) bioassay studies and the presence of nitrogen-fixing blue-green algae.

Long-Term Changes -- Average annual values for selected major phytoplankton nutrients were calculated for Muskegon Lake (see Table 26) and significant temporal trends were observed. Average surface water ammonia levels decreased 44% in the period following wastewater diversion while average surface nitrate concentrations increased 98%. Bottom water average ammonia concentrations decreased 53% and nitrate concentrations increased 62%. This resulted in a net increase in surface dissolved inorganic

Table 26. AVERAGE ANNUAL VALUES FOR SELECTED WATER QUALITY PARAMETERS IN MUSKEGON LAKE, 1972-1975

Year	Ammonia (μg N/ℓ)	Nitrate (μg N/ℓ)	Dissolved Inorganic Nitrogen (μg N/ℓ)	Total Dissolved Phosphorus (μg P/ℓ)	Total Phosphorus (μg P/ℓ)	Soluble Reactive Phosphorus (μg P/ℓ)	Chlorophyll *a* (μg/ℓ)	Secchi Disc (m)
				Surface				
1972	68	68	136	33	67	20	25.2	1.47
1973	42	64	106	20	58	7	16.5	1.53
1974	37	91	128	13	37	5	11.0	1.57
1975	38	135	173	15	42	4	9.5	1.58
				Bottom				
1972	237	90	327	44	77	13		
1973	141	77	218	25	73	15		
1974	146	111	257	16	42	7		
1975	111	146	257	14	45	4		

nitrogen and a net decrease in same in the bottom waters. Analysis of river loadings indicates that the changes in dissolved inorganic nitrogen reflect an increase in oxidation. This may result either from improved nitrification of the ammonia in wastewater aeration lagoons or by soil microorganisms on the disposal site. The net result of this would be favorable as a reduction in nitrogenous biochemical oxygen demand in the lake should be realized. A reduction in oxygen demand could reduce the duration of hypolimnetic anoxia and ultimately lessen phosphorus flux from the sediments. Analysis of the lake dissolved oxygen data show a possible improvement in hypolimnetic dissolved oxygen concentrations (see Figure 68).

A significant reduction in phosphorus concentrations was observed. Total dissolved and soluble reactive phosphorus were reduced 55 and 80% in the surface waters and 68 and 69% in the bottom waters. The reduction in phosphorus concentrations is supported by loading data. Such reductions may be the result of the ability of soils (at the disposal site) to retain phosphorus, a process not applicable to inorganic nitrogen. These levels of dissolved phosphorus are approaching the point where they may govern phytoplankton production.

No significant changes were observed in primary productivity levels in the period following diversion. Significant reductions in chlorophyll *a* levels were noted, although this had limited effect on water clarity (Secchi disc). This observation is supported by data from studies by Dillon and Rigler (1974 and 1975). As shown in Figure 76 Secchi disc is insensitive to fractional changes in chlorophyll *a* at the levels observed in Muskegon Lake. The system has, however, moved to a condition which is more sensitive to further change. The chlorophyll *a* reduction required to increase the average summer Secchi disc reading by one meter would be an additional 50 to 70% over 1975 values (see Figure 76). The total phosphorus reduction necessary (see Figure 77) would be approximately 40 to 60%.

Summary -- The reduction in nutrient load to Muskegon Lake as a result of the diversion project was expected to be 38% for total phosphorus and 48% for dissolved inorganic nitrogen. The actual observed load reductions were only 20 to 25% for total phosphorus and inorganic nitrogen because upstream loads had increased. The average nutrient and water quality conditions for Muskegon Lake for 1972 and 1975 are presented in Figure 78 along with data from Lakes Michigan and Erie for comparison.

Muskegon Lake data on nutrients, chlorophyll *a*, and Secchi disc reveal that significant reductions, particularly in phosphorus were achieved. This resulted in a large reduction in chlorophyll *a*. The reduction in chlorophyll *a* was not reflected in water clarity. The system has, however, been shifted into a more sensitive condition where further changes will increase Secchi disc depths. The change in chlorophyll *a* coupled with the expected reduction in biological oxygen demand load has resulted in increased levels of dissolved oxygen in the hypolimnion.

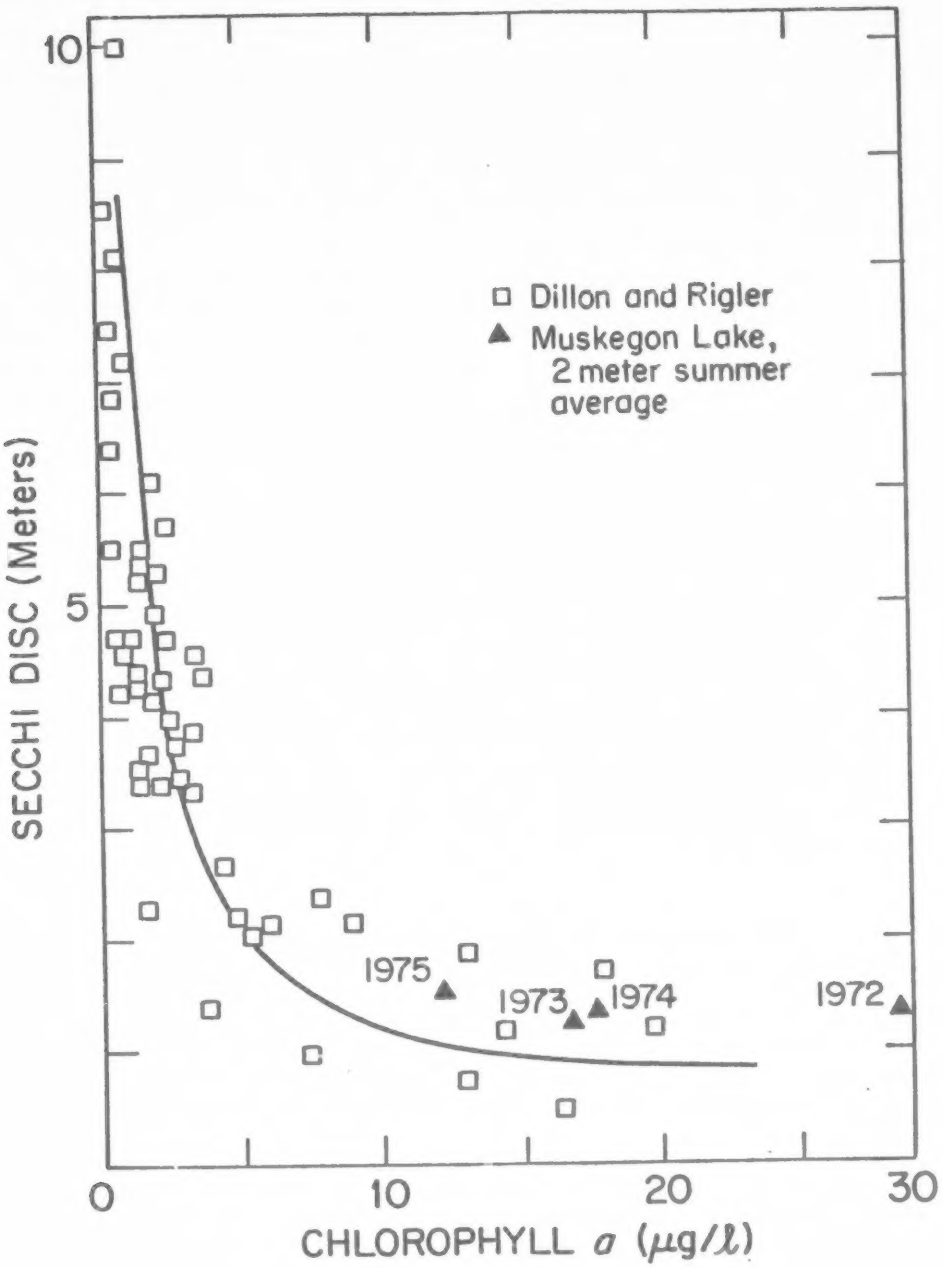


Figure 76. Correlation between chlorophyll *a* and Secchi disc (after Dillon and Rigler, 1975).

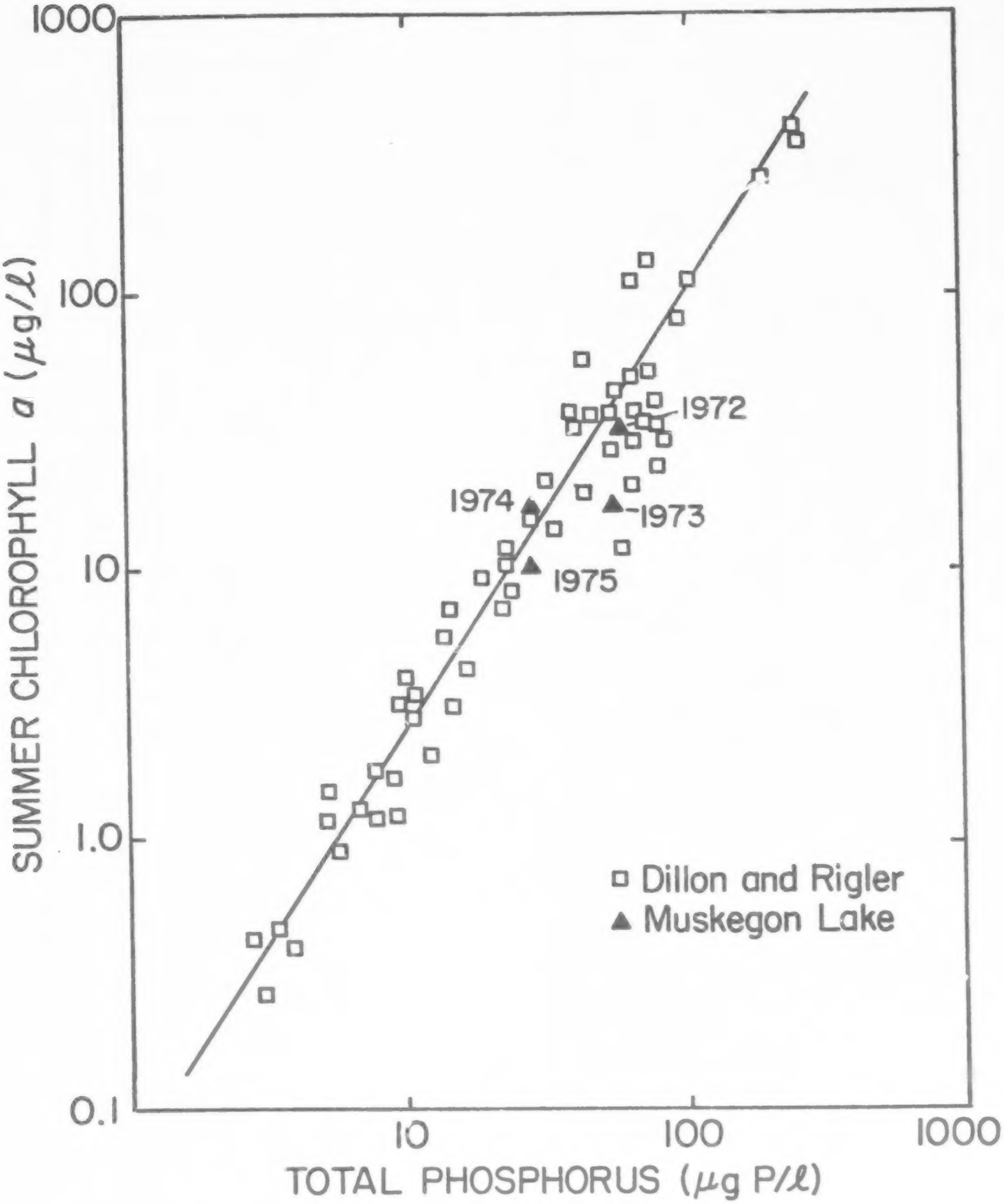


Figure 77. Correlation between phosphorus and summer chlorophyll *a* (after Dillon and Rigler, 1974).

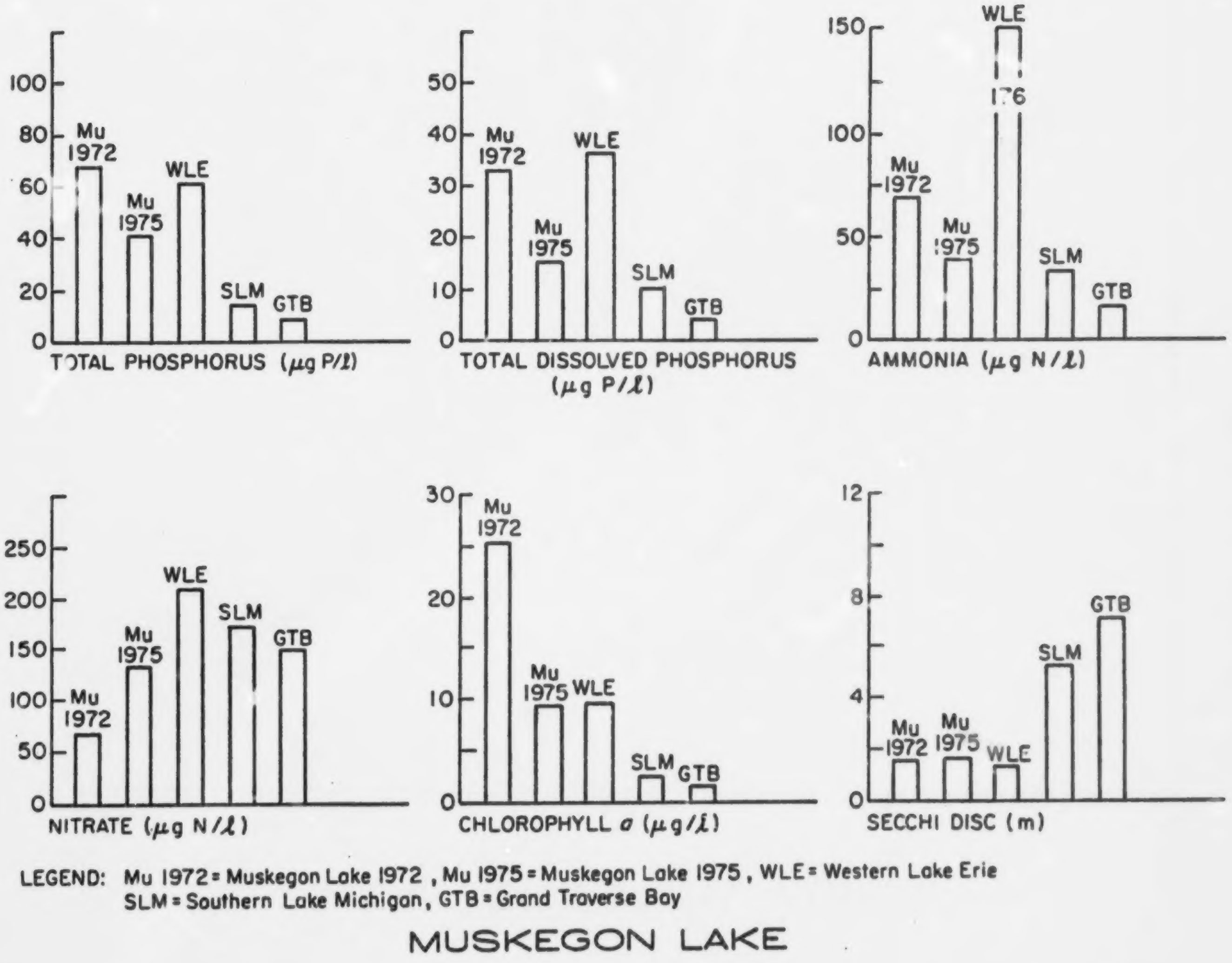


Figure 78. Pre- and post-diversion status of selected two-meter, yearly average water quality parameters in Muskegon Lake.

SECTION X

MONA LAKE

INTRODUCTION

Mona Lake is the smallest and southernmost of the three study lakes (see Figure 6). It is a very narrow, riverine lake, with a length of about 6.5 kilometers and a maximum width of 0.5 kilometers. The lake covers an area of 2.8 km^2 and has a mean depth of 4.1 meters and a volume of 1.15×10^7 m^3. The maximum lake depth is 12.8 meters (USEPA, 1975d). The lake has very limited shallow littoral areas due to rapid shoreline drop off.

The lake receives flow from several tributaries of which Black Creek and Little Black Creek are the most significant. Mona Lake is connected to Lake Michigan via a small channel. The channel is barely navigable, even by small pleasure boats.

The area surrounding Mona Lake is primarily residential. The lake is influenced by the urbanized area of Muskegon Heights. Little Black Creek flows through this area. Numerous industries and the Muskegon Heights Wastewater Treatment Plant have discharged to the lake by this creek. Detailed information on these discharges is given in Appendix F.

TRIBUTARY-RELATED CONSIDERATIONS

Hydrology -- The Mona Lake drainage basin is the smallest of the Muskegon Lakes' basins with an area of 212.4 km^2. The entire basin is within Muskegon County. Although several small creeks empty into the lake, only Black Creek and Little Black Creek are of sufficient size to merit monitoring. Black Creek is the largest tributary to Mona Lake; it drains 159.5 km^2 and discharges an average flow of 1.28 m^3/sec (see Table 27). Little Black Creek drains 15.5 km^2 and discharges an average flow of 0.15 m^3/sec. The remainder of the drainage, 34.6 km^2, is estimated to contribute a yearly average flow of 0.30 m^3/sec. The average tributary discharge flows given in Table 27 were extrapolated by USGS from data collected on 14 dates during 1972 and 1973. Average flows during the data collection period were nearly double the reported long-term average.

Information regarding the daily flow in Black and Little Black Creeks has not been gathered regularly by the USGS. No regular USGS gauging stations exist on rivers in the Mona Lake basin. To calculate the Black Creek flows on specific dates during this investigation, USEPA data were correlated with USGS flow measurements of the Muskegon River at Newaygo. A factor was then used to convert Muskegon River USGS gauged flow to an estimate for Black Creek flow on a specific date. The factor used was 0.025. A similar correlation for Little Black Creek was developed using

Table 27. MONA LAKE DRAINAGE BASIN CHARACTERISTICS
(PROJECTED LONG-TERM AVERAGE)

Description	Area (km^2)	Percentage of Total	Discharge (m^3/sec)	Percentage of Total
Black Creek	159.5	75.1%	1.28	74.1%
Little Black Creek	15.5	7.3%	0.15	8.6%
Miscellaneous Tributaries & Direct Drainage	34.6	16.3%	0.30	17.3%
Lake Surface Area	2.8	1.3%		
Total	212.4	100.0%	1.73	100.0%

(Modified from USEPA, 1975d).

the Black Lake Creek USGS gauged flow. A factor of 0.769 was derived. The precision and accuracy of these factors is complicated by numerous variable point discharges in the Little Black Creek basin and by the drainage effects of the spray-irrigation site. Corrections for these effects are extremely difficult and were not attempted. The accuracy of the correlation is sufficient for evaluation of relative flows and nutrient loads.

The mean hydraulic retention time for Mona Lake is 76 days (USEPA, 1975d). This implies that the average hydrologic tributary input to Mona Lake is sufficient to exchange the total water volume in the lake about five times each year. This calculation does not account for seasonal variability or incomplete mixing in the lake.

Concentrations of Chemical Species in Tributaries -- The concentration of various water quality parameters was measured in Black Creek and Little Black Creek. No other tributary or drainage sources were considered large enough to monitor. A summary of the water quality findings is given in Table 28. The concentrations measured in the two creeks were significantly different and, as such, merit individual discussion.

The water quality observed at the Black Creek mouth was distinctly different from that observed in the Muskegon and White Rivers. Measured concentrations of phosphorus and dissolved silicon were similar to the other rivers; however, nitrogen, iron, and chloride concentrations were much higher. The total study period averages at the Black Creek mouth were 50.8 μg P/ℓ, 3.5 μg Si/ℓ, 260 μg NH_3-N/ℓ, 651 μg NO_3-N/ℓ, 746 μg Fe/ℓ, and 35.9 μg Cl/ℓ. The elevated nitrate, iron, and chloride concentrations were believed to be an influence of groundwater and non-point sources. Black Creek is a much smaller tributary than the Muskegon and White Rivers and is suspected to be dominated more by groundwater infiltration. Groundwater generally has relatively high nitrate and iron concentrations compared with surface waters. This is a consequence of a number of factors including ammonification, nitrification, and redox potential. The high chloride in the water could also originate from groundwater or urban runoff. In addition, Black Creek receives seepage from industrial waste lagoons.

The influence of groundwater on Black Creek is also observed in plots of the nutrient concentrations measured at Station 27 (Figures 79 and 80). The seasonal variation in chemical concentrations in Black Creek was less than that observed for the Muskegon and White Rivers. Figures 81 to 84 are plots of concentrations of various parameters measured at stations in Black Creek.

During the period of this investigation no significant or conclusive trends in Black Creek water quality were observed. Major improvements in water quality were not expected because there were no diversions of municipal or industrial discharges from Black Creek. The only industrial

Table 28. AVERAGE CONCENTRATIONS OF SELECTED CHEMICAL SPECIES IN TRIBUTARIES TO MONA LAKE, 1972-1975

Tributary	Year	Chloride (mg Cl/ℓ)	Ammonia (μg N/ℓ)	Nitrate (μg N/ℓ)	Soluble Reactive Phosphorus (μg P/ℓ)	Total Dissolved Phosphorus (μg P/ℓ)	Total Phosphorus (μg P/ℓ)	Silicon (mg Si/ℓ)	Dissolved Iron (μg Fe/ℓ)	Total Iron (μg Fe/ℓ)
Black Creek (at mouth) (Station 27)	1973	36.9	272	650	10.5	25	73	3.2	246	796
	1974	34.2	289	646	11.7	19	37	3.6	152	753
	1975	36.9	207	659	9.3	18	42	3.5	162	695
	1973-1975	35.9	260	651	10.6	21	51	3.5	182	746
Black Creek (Station 2)	1972	16.6	113	811	--	11	20	4.4	--	--
	1973	11.0	107	453	4.9	11	49	3.6	105	1978
	1974	12.8	149	721	3.5	9	29	3.7	353	3018
	1975	18.9	195	719	2.6	10	27	3.7	205	2356
	1972-1975	14.2	141	651	3.7	10	34	3.8	232	2497
Black Creek (Station 25)	1973	11.7	54	600	4.7	15	32	3.3	123	902
	1974	12.4	93	785	5.7	12	35	3.4	126	1057
	1975	10.6	102	617	9.7	18	41	3.1	170	871
	1973-1975	11.6	81	650	6.5	15	35	3.3	138	952
Little Black Creek (Station 26)	1973	69.9	2608	5716	530	683	1022	3.7	101	1245
	1974	86.7	630	5811	588	585	922	2.9	44	509
	1975	67.0	436	4538	87	101	234	2.6	57	561
	1973-1975	75.0	1314	5424	429	489	755	3.1	66	786

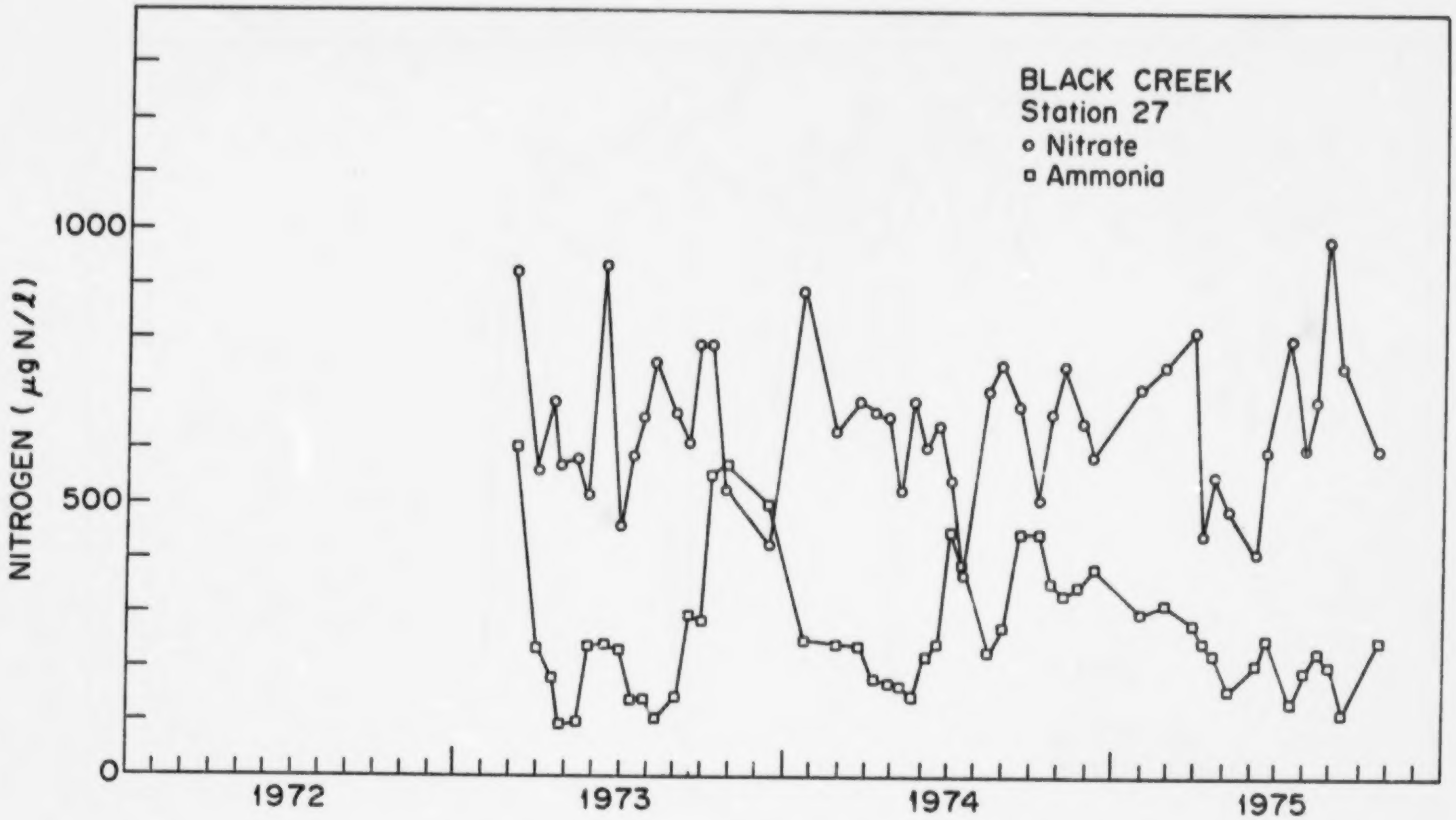


Figure 79. Dissolved inorganic nitrogen concentrations in Black Creek, 1973-1975.

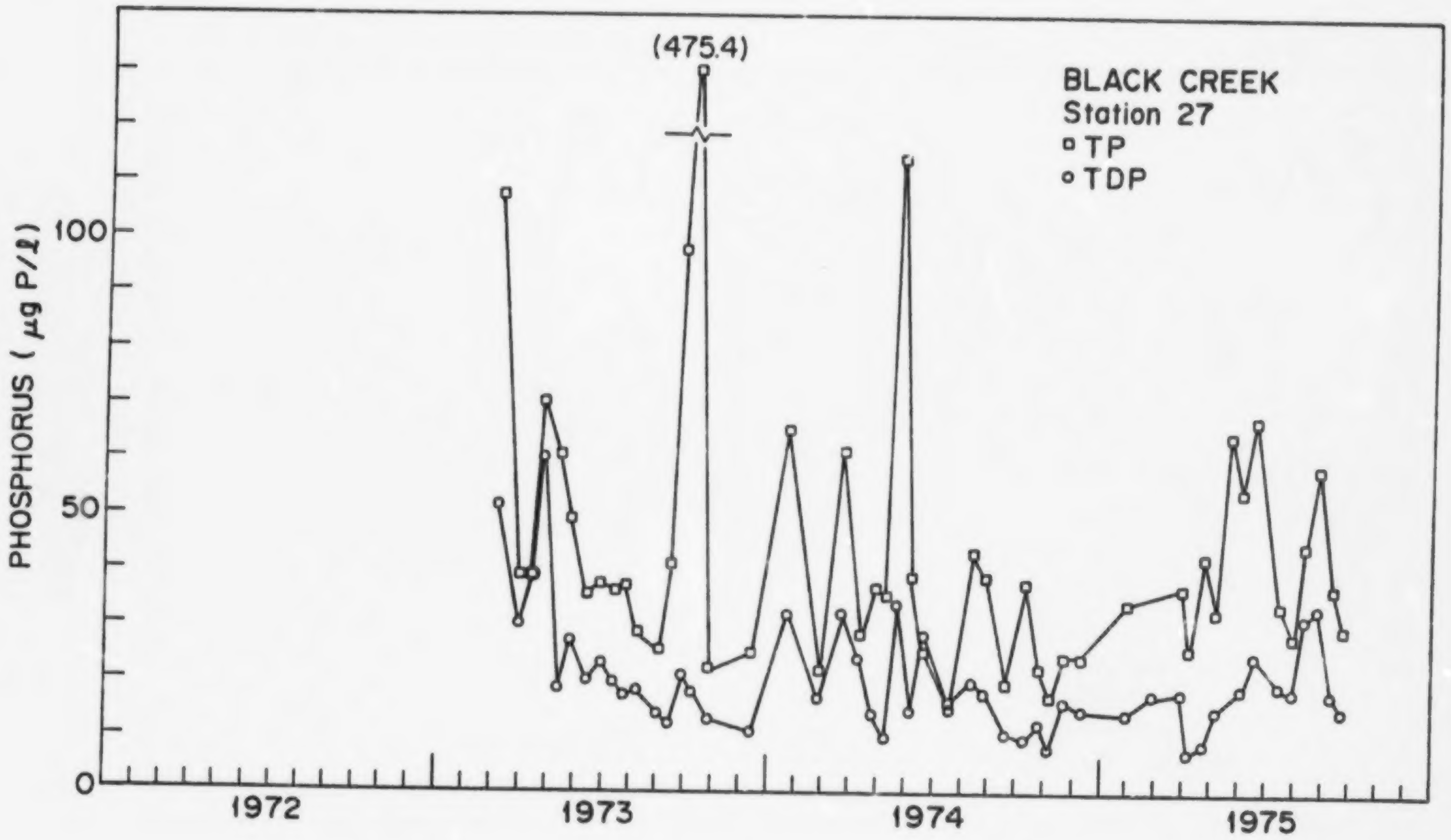


Figure 80. Phosphorus concentrations in Black Creek, 1973-1975.

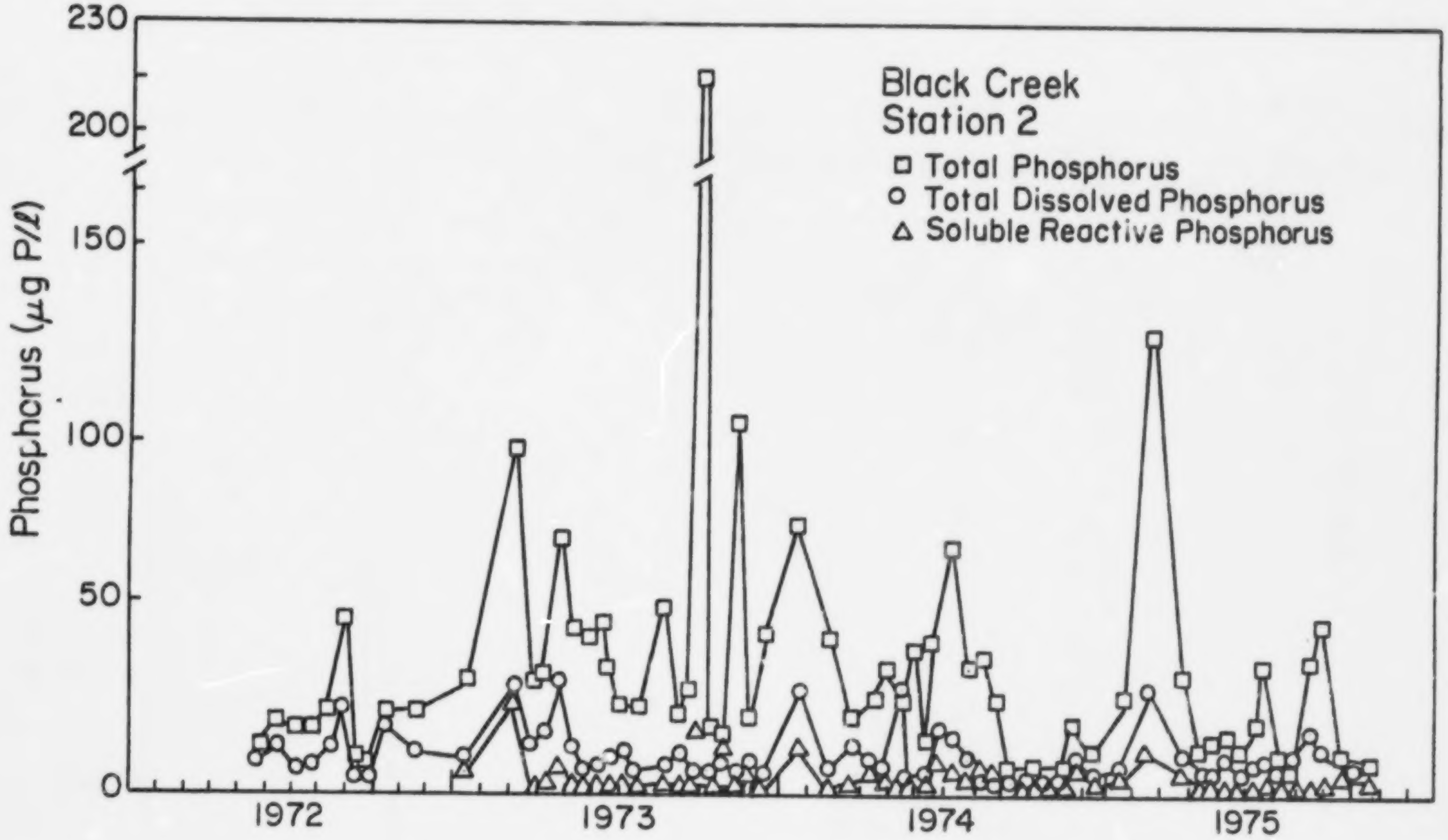


Figure 81. Phosphorus concentrations in Black Creek, 1972-1975.

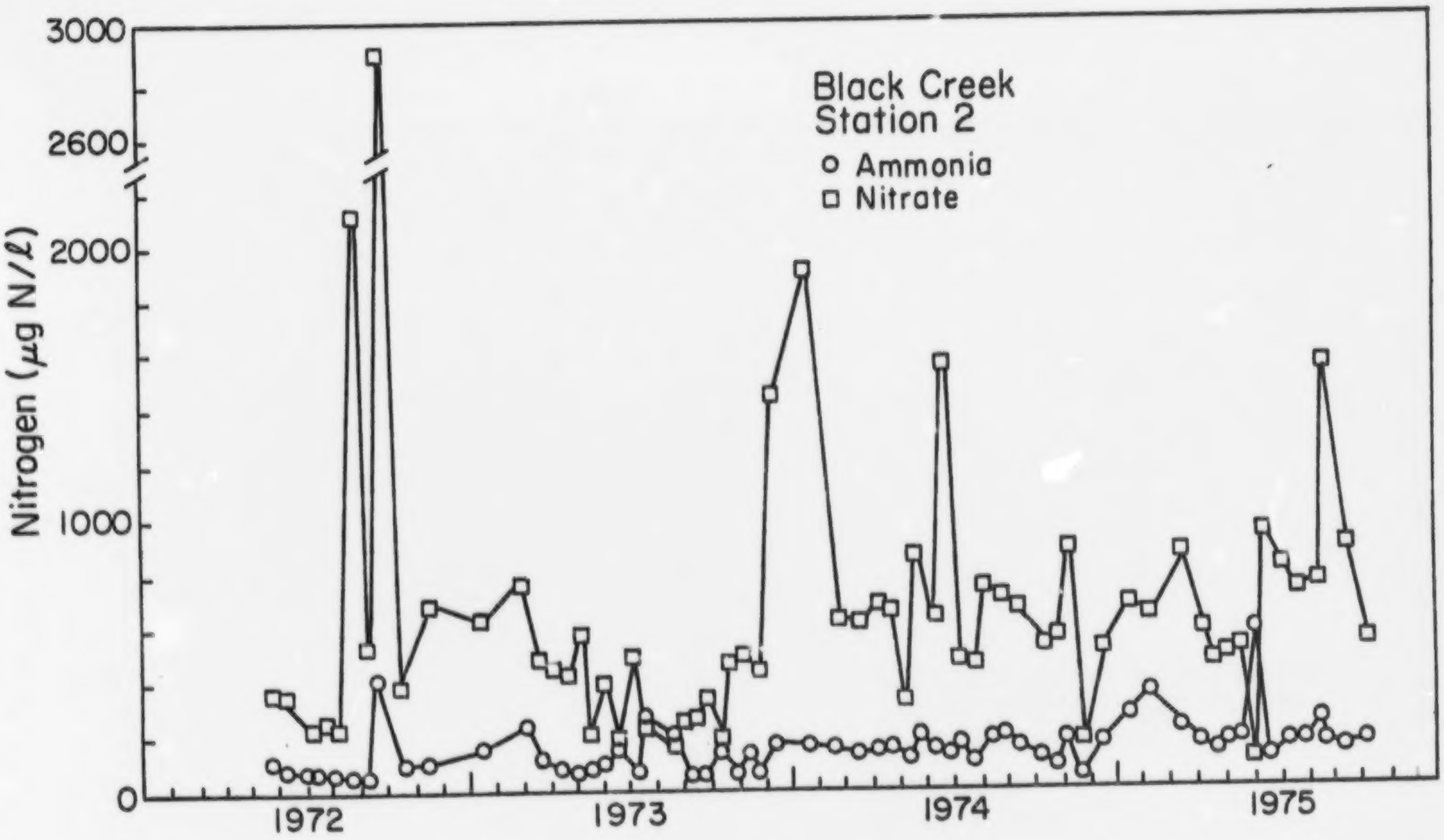


Figure 82. Dissolved inorganic nitrogen concentrations in Black Creek, 1972-1975.

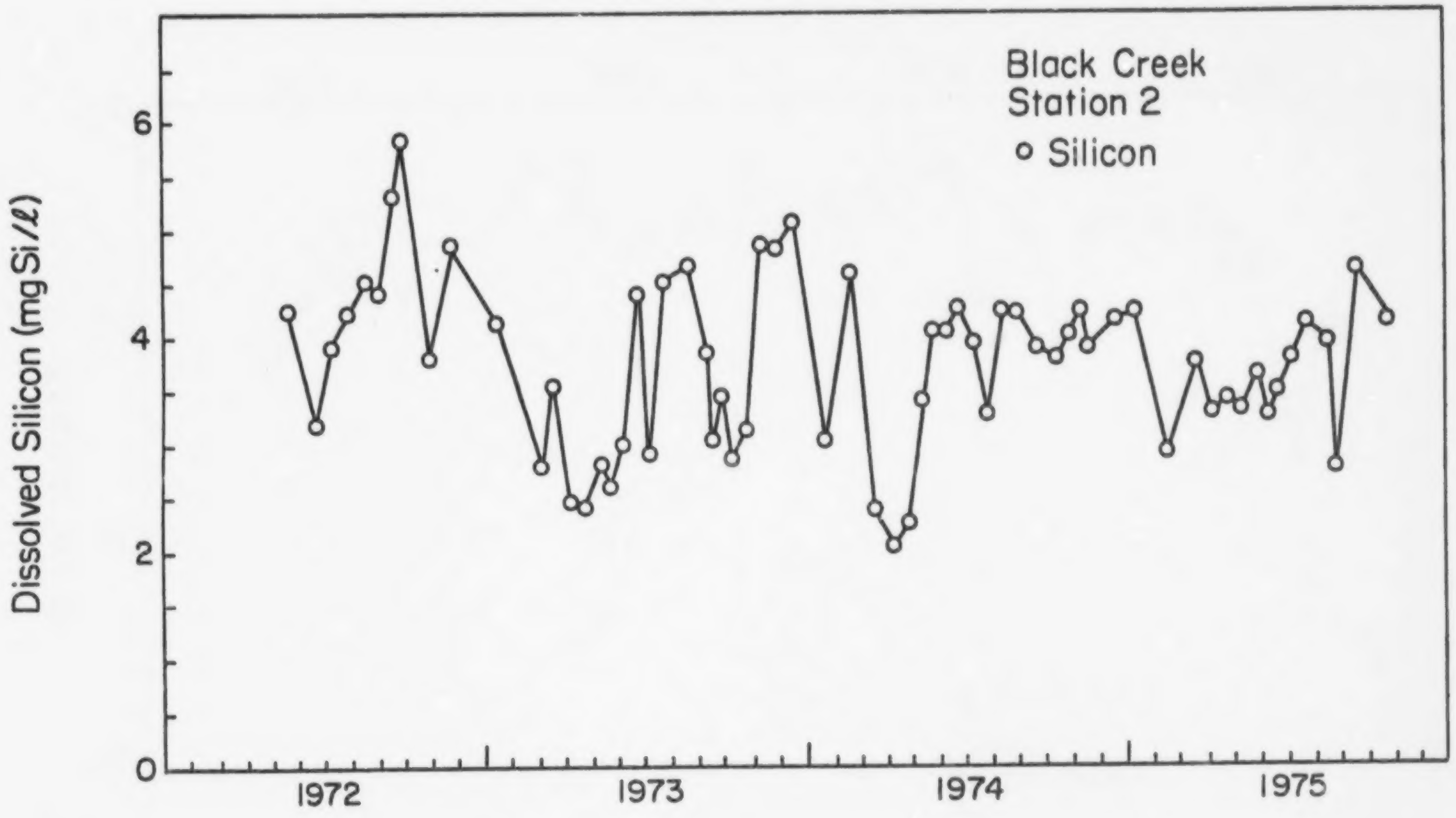


Figure 83. Dissolved silicon concentrations in Black Creek, 1972-1975.

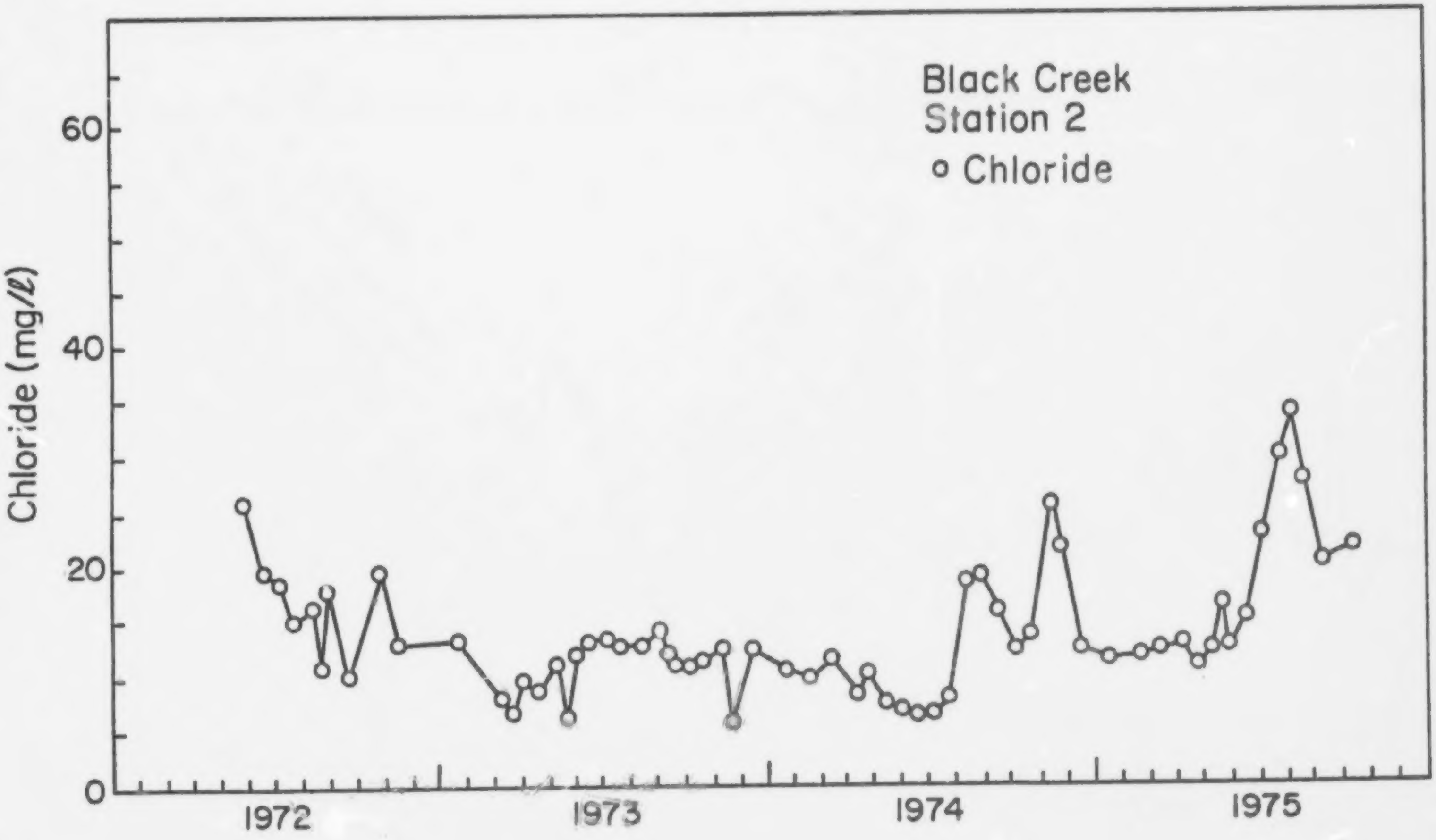


Figure 84. Chloride concentrations in Black Creek, 1972-1975.

or municipal contamination that Black Creek is known to receive originates from industrial waste lagoon seepage. Black Creek also receives drainage from the spray-irrigation site. One possible temporal trend was observed for phosphorus. Phosphorus concentrations at both up and downstream stations were observed to be higher in 1973. This was not a consequence of the diversion project. No changes in Black Creek water quality were observed to result from spray-site drainage.

An evaluation of the spatial variation in Black Creek water quality can provide more information on the relative influence of sources which contribute materials to the river. Station 27 represents water quality at the mouth of Black Creek. Station 2 represents upstream water quality and includes the effects of the spray-irrigation drainage. Station 25 represents the water quality of the headwaters of Black Creek and is a measure of the effect of the spray-irrigation drainage closer to its entry to the river. Sample collection upstream of the spray site was not possible because of the limits of the drainage basin. As a consequence, the influence of the spray-site drainage could not be observed directly.

Observations of upstream water quality reveal the strong influence of groundwater. Iron and nitrate concentrations at Station 25 were very high (approximately 1,000 μg Fe/ℓ and 650 μg N/ℓ). As the water moved downstream however, iron concentrations increased at Station 2 and then dropped at Station 27. This is likely associated with non-point sources, dilution, oxidation and precipitation. Ammonia concentrations increased with distance downstream with an associated increase in ammonia concentrations. This may be a result of biologic activity or additional contributions of ammonia. Phosphorus concentrations were noted to increase by 25 to 50% downstream. This is a result of the increased influence of runoff and possibly seepage from industrial lagoons. Chloride concentrations also increased with distance downstream again suggesting groundwater infiltration, salt deposits, urban runoff, or lagoon seepage.

Concentrations in Little Black Creek were distinctly different from that observed in Black Creek. During 1973 (the only year including any observations on Little Black Creek prior to diversion), the concentration of phosphorus, nitrogen, and chloride were all observed to be much higher than those measured in Black Creek. Chlorides were approximately twice as high while phosphorus and nitrogen were an order of magnitude higher. The average total phosphorus concentration was 1,022 μg P/ℓ while the ammonia and nitrate concentrations were 2,608 μg N/ℓ and 5716 μg N/ℓ respectively (see Figures 85 and 86). Little Black Creek flows through a high industrialized area of Muskegon and, prior to diversion, received numerous industrial and municipal waste discharges as well as urban runoff and storm sewer drainage. This influence will be discussed in more detail in the following section on nutrient loads. The influence

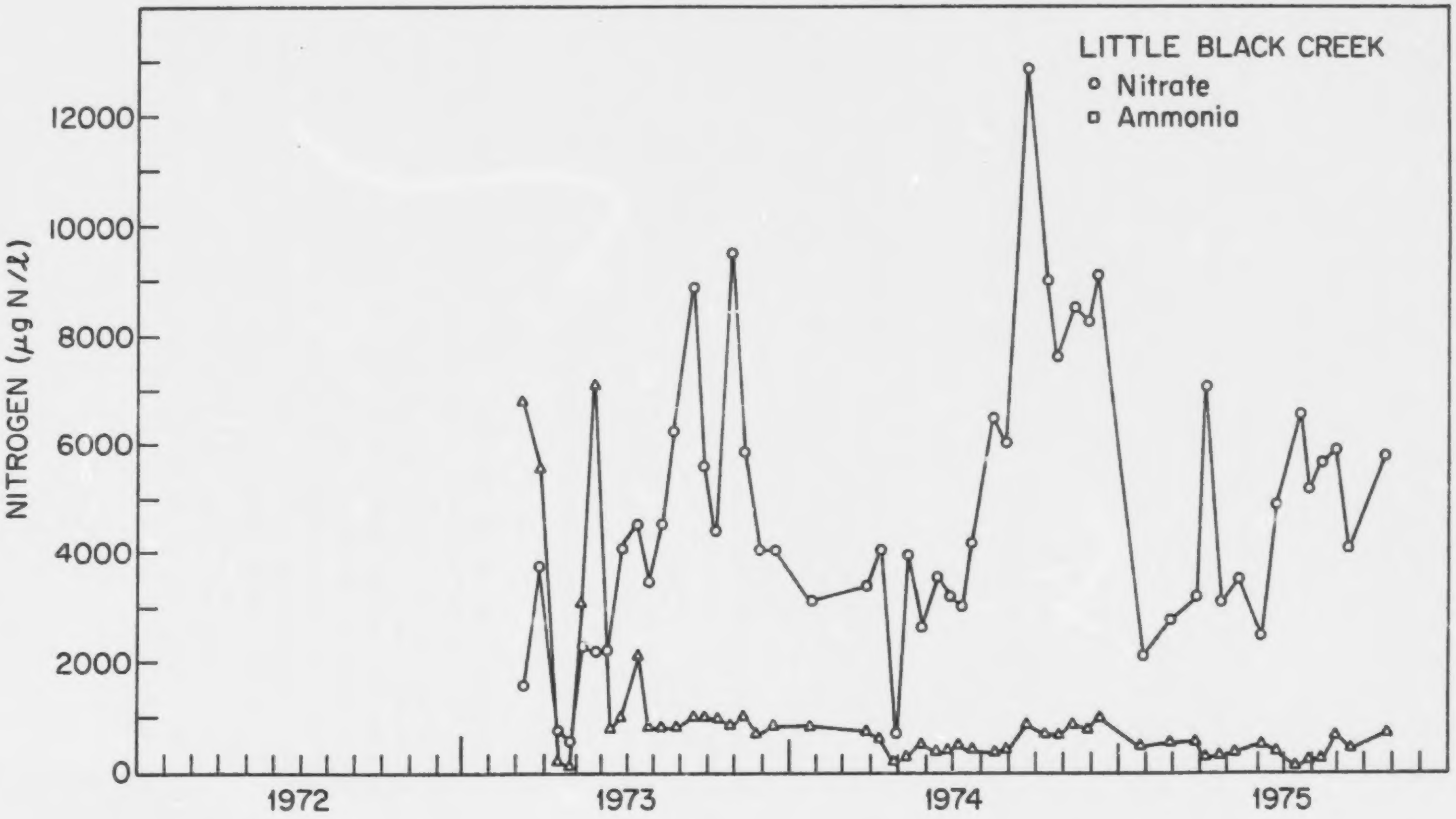


Figure 85. Dissolved inorganic nitrogen concentrations in Little Black Creek, 1973-1975.

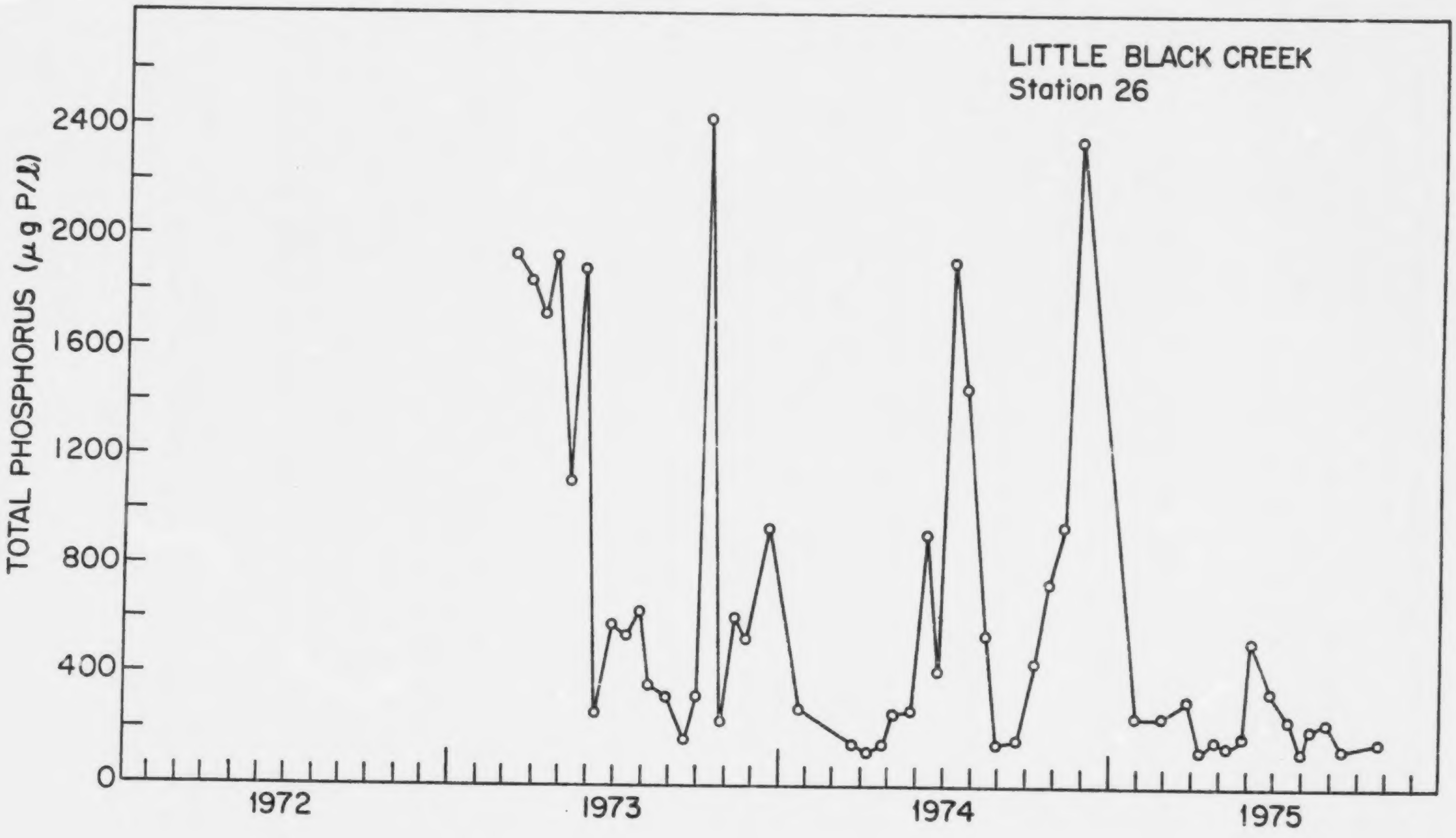


Figure 86. Total phosphorus concentrations in Little Black Creek, 1973-1975.

of these discharges can be readily seen in the data for 1974 and 1975. During this period the Muskegon Heights Wastewater Treatment Plant was diverted and other industrial discharges reduced their nutrient input to the creek. Concentrations of nutrients were much less in 1975; this was particularly true for phosphorus and iron. No data on upstream water quality or river mouth water quality were available prior to 1973 to accurately assess the specific influences of the various municipal and industrial discharges.

Nutrient Loads -- The primary sources of nutrients to Mona Lake are Black and Little Black Creeks. Municipal and industrial discharges have contributed significantly to the nutrient budget of the lake, although none discharged directly to the lake, instead Little Black Creek has received these discharges. The location of these industries is presented in Figure 87. Other data on these discharges is given in Appendix F. Other nutrient sources such as minor tributaries, septic tank seepage, and atmospheric precipitation are difficult to quantify but were considered minor (USEPA, 1975d).

The tributary nutrient loads to Mona Lake from Black Creek and Little Black Creek were calculated by multiplying measured concentrations by flows. The measured concentrations from Little Black Creek were below the locations of point discharges, urban runoff, and storm runoff. The water quality measurements, therefore, included the effects from these sources. Measurements of Black Creek water quality at Station 27 included the effects of waste lagoon seepage. All flow estimates were based on limited direct measurements but are acceptable for establishing reasonable estimates of the nutrient loads. A summary of the yearly tributary nutrient loads is given in Table 29. Plots of nitrogen and phosphorus loads to Mona Lake from Black and Little Black Creek are presented in Figures 88 through 91. No direct monitoring of Little Black Creek water quality was conducted in 1972.

Nutrient loads from urban storm runoff were suspected to be significant although direct monitoring of these sources was beyond the study scope. The Western Michigan Shoreline Federal 208 Planning Agency has roughly estimated (without site specific data) the contribution of total phosphorus from storm runoff to Mona Lake to be 4.4 thousand kg P/year (Weaver, 1976). This represents a significant fraction of the total load. However, over 75% of the storm runoff drains into Little Black Creek above Station 26. Tributary monitoring at Station 26 was conducted in both clear and rainy conditions; therefore, this source was included in the tributary monitoring.

Estimates of the atmospheric nutrient inputs to Mona Lake can be calculated using data from Johnson (1976) for Saginaw Bay. These data show that calculated total phosphorus atmospheric inputs to Mona Lake would be much less than 1% of the total load.

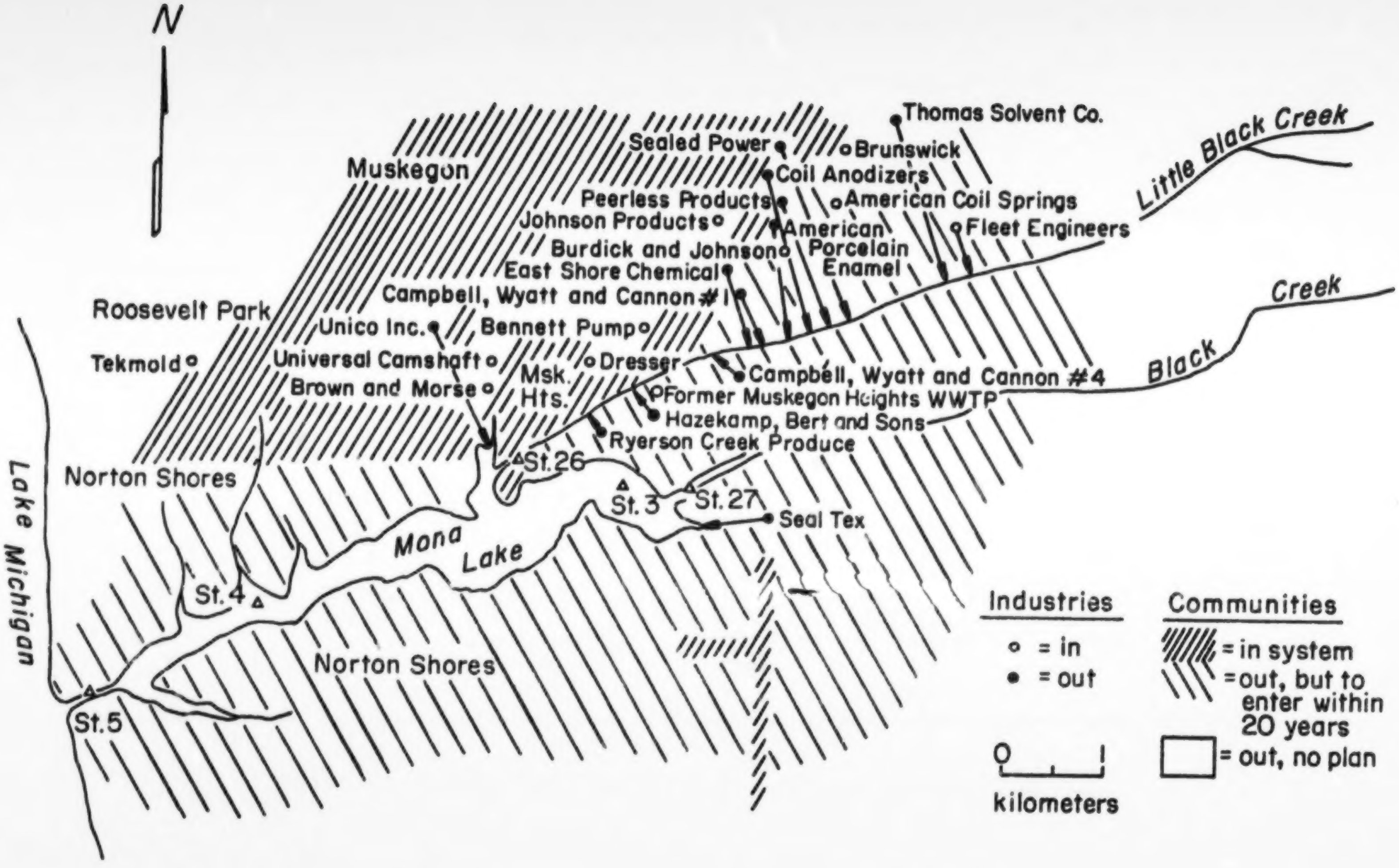

N
Muskegon
Roosevelt Park
Tekmold
Norton Shores
Norton Shores
Lake Michigan
Mona Lake
Msk. Hts.
Sealed Power
Brunswick
Thomas Solvent Co.
Coil Anodizers
Peerless Products
American Coil Springs
Johnson Products
American Porcelain Enamel
Fleet Engineers
Burdick and Johnson
East Shore Chemical
Campbell, Wyatt and Cannon #1
Unico Inc.
Bennett Pump
Universal Camshaft
Dresser
Brown and Morse
Campbell, Wyatt and Cannon #4
Former Muskegon Heights WWTP
Hazekamp, Bert and Sons
Ryerson Creek Produce
Seal Tex
Little Black Creek
Black Creek
St. 26
St. 3
St. 27
St. 4
St. 5
Industries
○ = in
● = out
0 1
kilometers
Communities
= in system
= out, but to enter within 20 years
= out, no plan

Figure 87. Location and diversion status of point sources and municipalities in the Mona Lake basin.

Table 29. AVERAGE MONA LAKE TRIBUTARY LOADS IN KILOGRAMS PER DAY

Station	Year	Total Dissolved Phosphorus as P	Total Phosphorus as P	Nitrate + Ammonia as N	Silicon as Si	Chloride as Cl
Black Creek (Station 27)	1973	3.6	9.1	121	379	4319
	1974	3.3	6.9	140	505	4835
	1975	3.0	7.6	154	630	6066
Black Creek (Station 2)	1972	0.5	1.1	55.5	212	737
	1973	0.7	2.9	37.2	215	658
	1974	0.8	2.6	73.2	283	901
	1975	0.7	2.5	83.1	295	1543
Little Black Creek (Station 26)	1973	23.5	31.9	131	66.3	1355
	1974	7.8	14.2	133	74.3	2544
	1975	4.0	8.6	183	91.4	2295

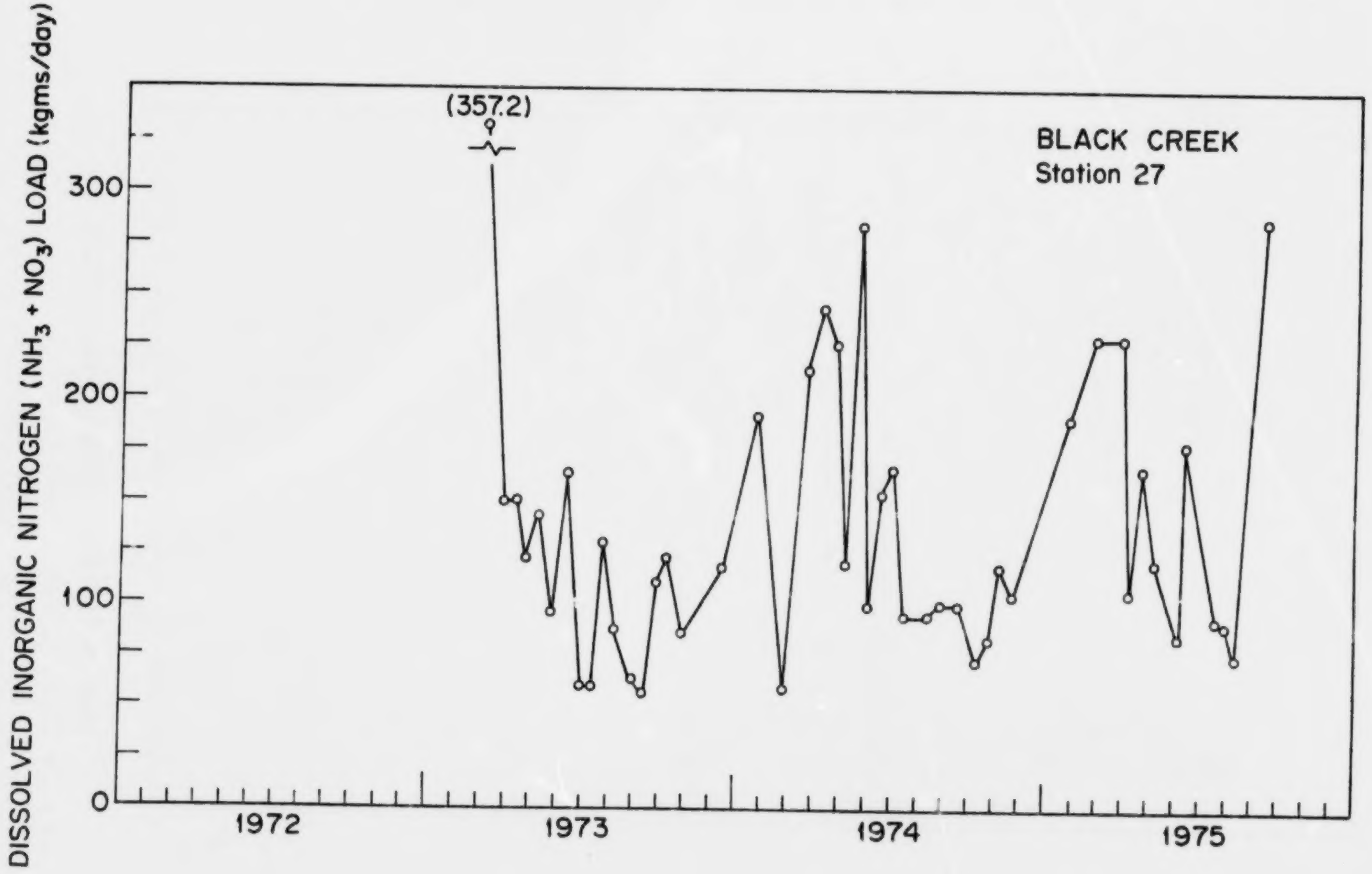


Figure 88. Dissolved inorganic nitrogen load of Black Creek, 1973-1975.

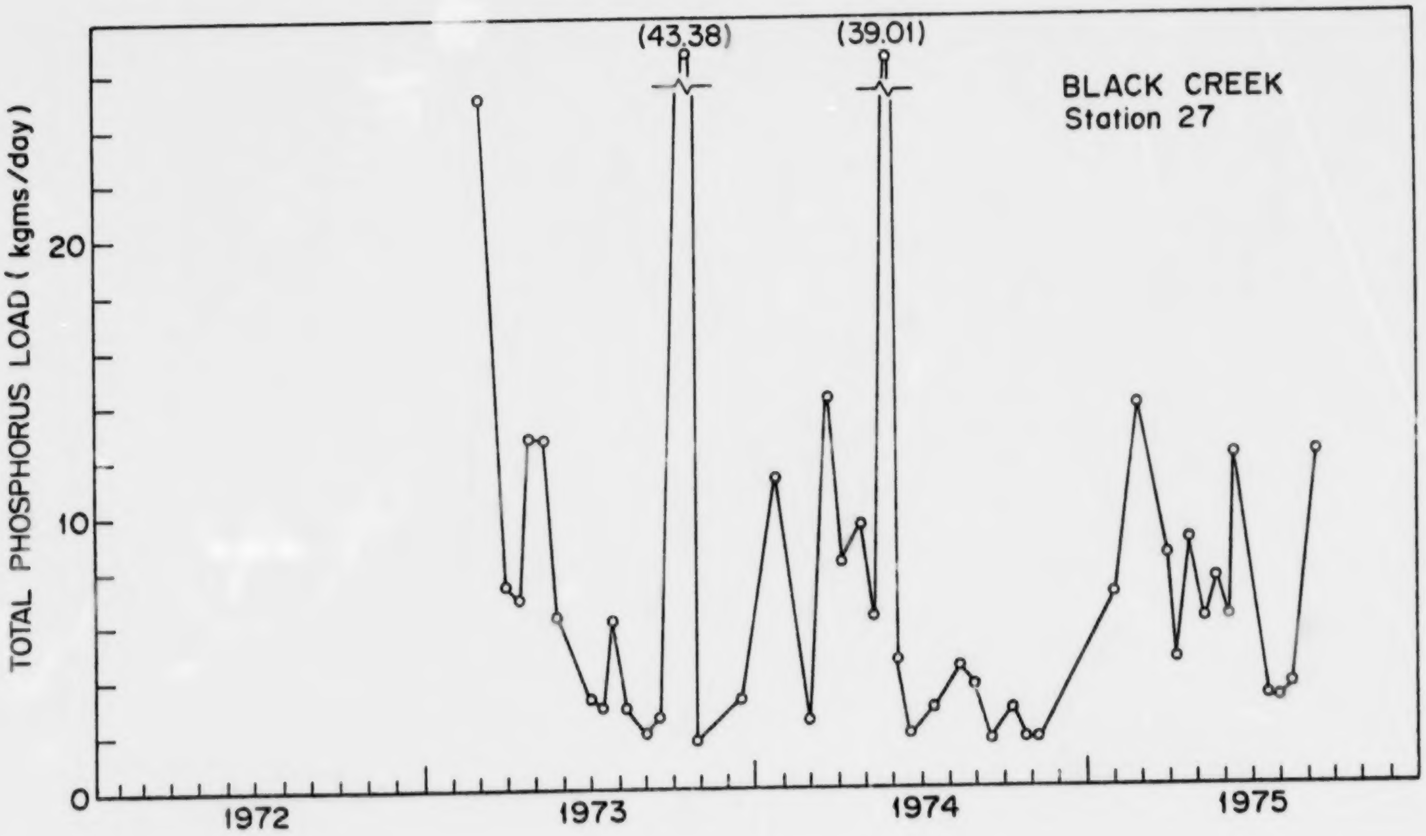


Figure 89. Total phosphorus load of Black Creek, 1973-1975.

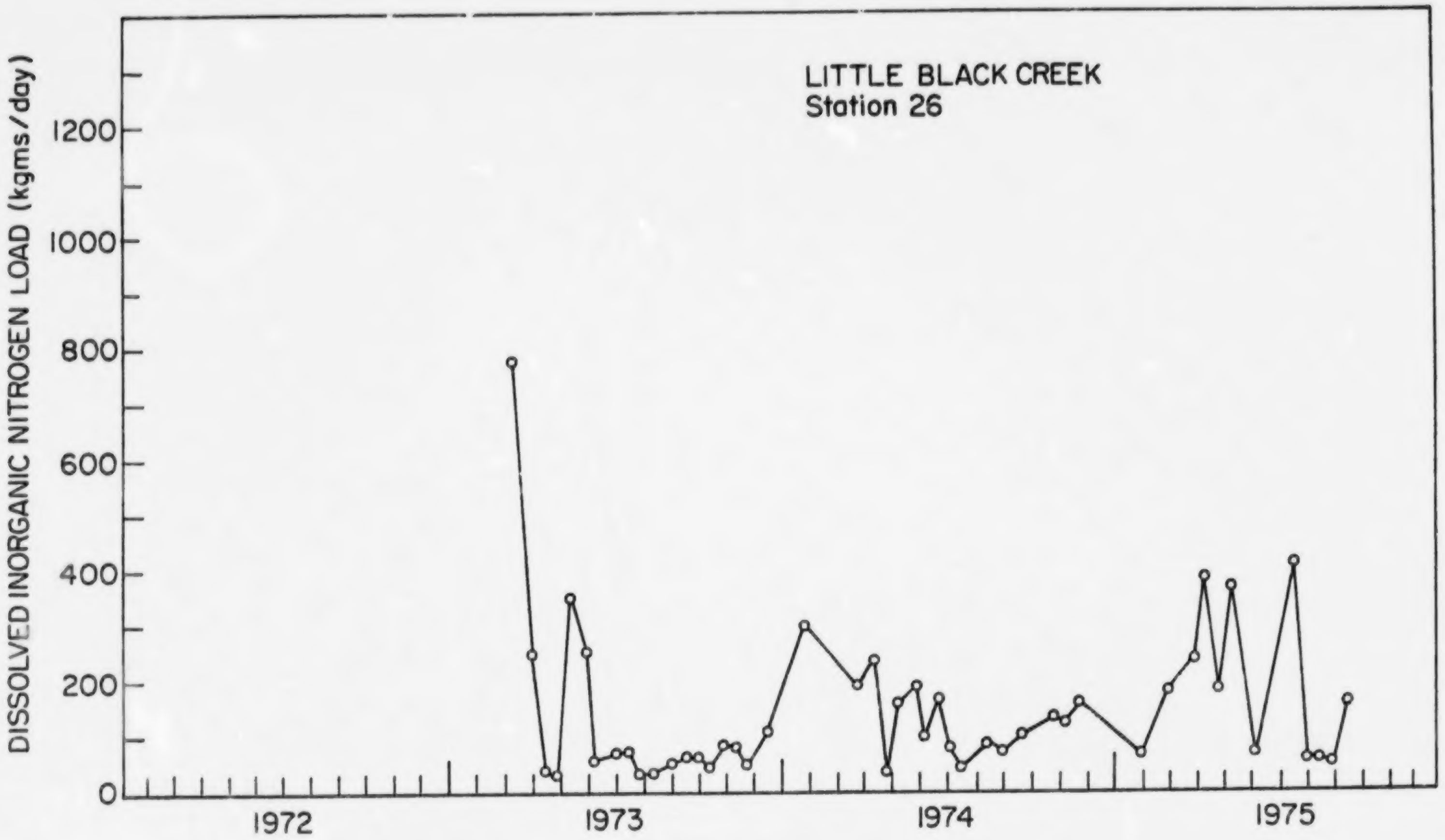


Figure 90. Dissolved inorganic nitrogen load of Little Black Creek, 1973-1975.

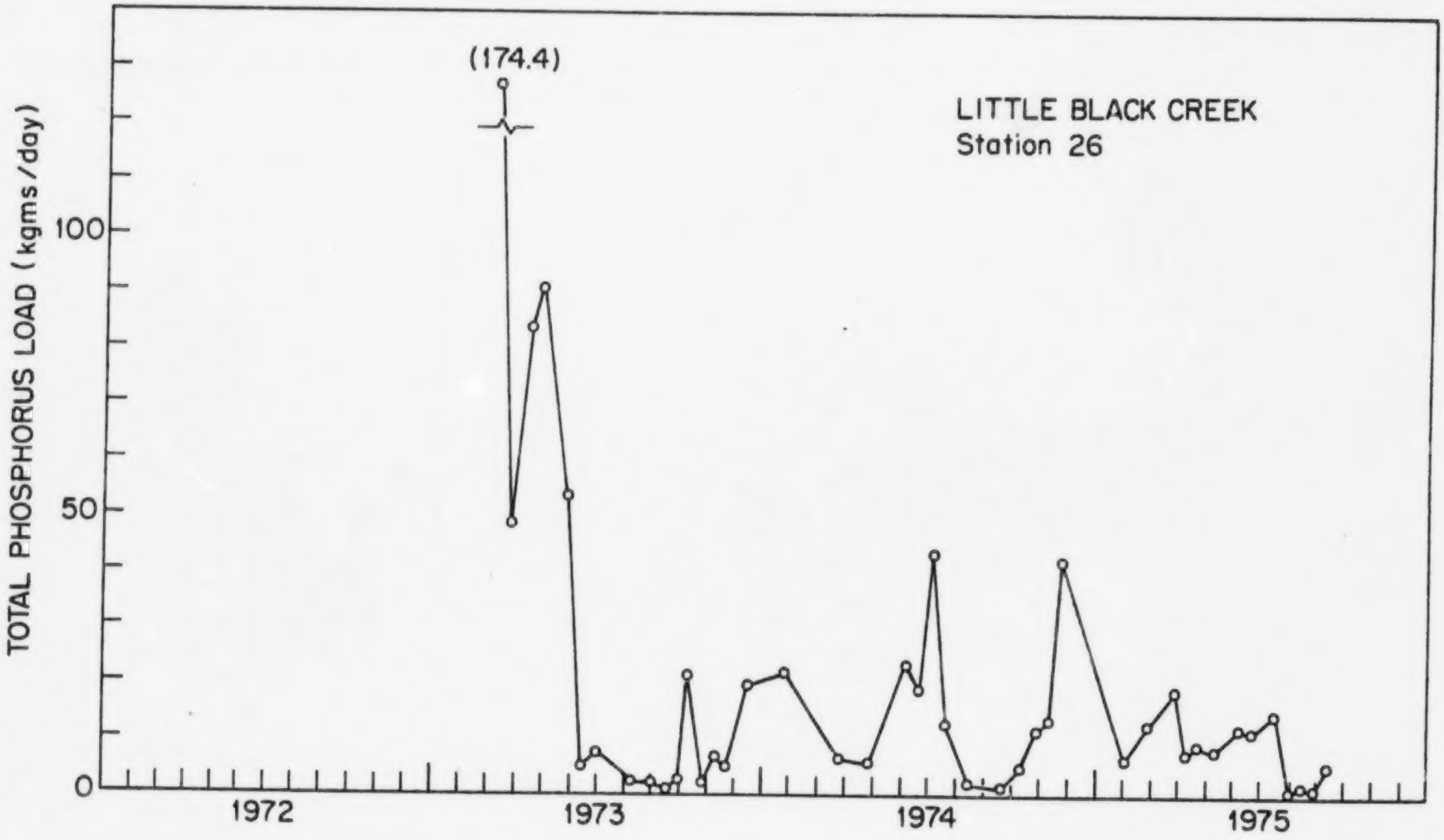


Figure 91. Total phosphorus load of Little Black Creek, 1973-1975.

An evaluation of specific municipal and industrial discharges is difficult. None are known to have discharged directly into Black Creek except for the drainage from the spray-irrigation site which began in 1974. Unquantified sources of contamination originate from industrial lagoon seepage. Little Black Creek flows through a highly industrialized area of Muskegon and receives numerous discharges from storm sewers, drains, etc., not all of which are monitored. Table 30 is a summary of the data available for discharges from some of these industries and municipalities. The important discharges are those of the Muskegon Heights Wastewater Treatment Plant and the Kersman Company (Coil Anodizers). The Muskegon Heights Wastewater Treatment Plant diverted its discharge in May of 1973. The Kersman Company discharges high loads of phosphorus to Little Black Creek and had not diverted its wastes during this study. Since December of 1974 the Kersman Company has maintained a much reduced phosphorus loading consistent with federal (NPDES) requirements. The specific data at which loads were reduced from the 1970 to 1975 level is not known because operating reports on discharge water quality are not available prior to December 1974. Information on the other industrial discharges for the study period is presented in Appendix F. Many of these discharges have plans to divert to the spray-irrigation system. Loading data was only available for major industrial discharges.

Yearly average nutrient loads to Mona Lake demonstrate a significant phosphorus reduction (approximately 60%) between 1973 and 1975. No 1973 to 1975 reduction in load was observed for nitrogen, silicon, or chloride. Measurements of the actual reduction since 1972 and a subsequent comparison between the measured and the anticipated reductions are difficult because of insufficient pre-diversion data. The original study plan did not include monitoring on Little Black Creek or the mouth of Black Creek. An estimate of the 1972 to 1975 reduction can be obtained from other available data.

The upstream Black Creek nutrient measurements are considered representative of non-point (non-urban) contamination. Upstream measurements do, however, include the effects of spray-irrigation drainage. This has been calculated as 0.44 kg P/day and 7.36 kg N/day for 1973; 0.76 kg P/day and 40.0 kg N/day for 1974; and 1.07 kg P/day and 52.4 kg N/day for 1975 (see Appendix E). The average of the upstream non-point loads from Black Creek are 1.8 kg P/day and 37 kg N/day. These loads were also used to estimate hypothetical Little Black Creek nutrient loads uneffected by urban runoff and industrial municipal discharges. The load was corrected based on a drainage area ratio. The resulting total non-point loads are for total phosphorus 2.0 kg/day and for dissolved inorganic nitrogen 41 kg N/day. These estimates do not include storm and urban runoff or waste lagoon seepage.

A comparison of Station 2 and Station 27 loading data indicate that on the average 5.2 kg P/day of total phosphorus and 75.0 kg N/day of dissolved inorganic nitrogen are contributed to Black Creek within this reach. This could result from waste lagoon seepage or other non-point sources such as storm runoff.

Table 30. ESTIMATED MUNICIPAL AND INDUSTRIAL LOADS TO MONA LAKE PRIOR TO DIVERSION IN KILOGRAMS PER DAY*

Source & Location of Discharge	Receiving Water	Chemical Oxygen Demand	Biochemical Oxygen Demand	Suspended Solids	Soluble Reactive Phosphorus as P	Total Phosphorus as P	Dissolved Inorganic Nitrogen as N	Total Nitrogen as N	Date of Diversion	Source
American Coil Spring Company, Muskegon	Keating Drain (Little Black Creek)	1.104	0.65	0.78	0.39	0.41	1.05	1.09	1/75	MWRC 1973
American Porcelain Enamel Company, Muskegon	Keating Drain (Little Black Creek)	3.66	0.81	6.28	0.005	0.02	0.06	0.12	--	MWRC 1973
Browne-Morse Company, Muskegon Heights	Merriam St. Sewer (Little Black Creek)	6.85	0.43	0.67	0.97	2.27	0.11	0.16	4/74	MWRC 1973
Campbell, Wyant, & Cannon Company, Plant #1, Muskegon Heights	Little Black Creek	5.90	—	53.1	0.01	0.08	1.77	>1.77	--	MWRC 1973
Campbell, Wyant, & Cannon Company, Plant #4, Muskegon Heights	Little Black Creek	2.54	--	2.22	0.005	0.01	>0.16	>0.16	—	MWRC 1973
East Shore Chemical Company, Muskegon	Little Black Creek	3.36	0.74	1.83	0.006	0.006	1.02	1.04	--	MWRC 1973
Johnson Products Company, Muskegon	Keating Drain (Little Black Creek)	0.56	0.70	0.84	0.005	0.01	0.07	0.24	74-75	MWRC 1973
Kersman Company, Coil Anodizers Muskegon**	Keating Drain (Little Black Creek)	--	--	--	24.7	35.44 1.59	>30.0	>30.0	--	MWRC 1973
Muskegon Heights Wastewater Treatment Plant, Muskegon Heights***	Little Black Creek	--	--	--	--	25.2 96.0	208 193	319 309	5/73	Muskegon Co.'75 Muskegon Co.'75
Sealed Power Corporation, Muskegon	Little Black Creek	5.83	1.94	49.5	0.05	0.02	0.74	1.14	--	MWRC 1973

*This table does not include industrial outfalls which prior to 1973 discharged to a municipal treatment system. It also does not include industries which discharge to private lagoons or septic systems. Nutrient loads to Mona Lake from these latter sources could not be directly quantified. Appendix F contains more information about other industrial outfalls.

**Treatment of wastes improved between 1970 and 1974.

***Two levels of discharge load represent plant operation with and without lime treatment.

The pre-diversion total industrial and municipal point loads are calculated to be 29.8 kg P/day of total phosphorus and 231 kg N/day of dissolved inorganic nitrogen. These were estimated assuming improved treatment of wastes for the Muskegon Heights Wastewater Treatment Plant and the Kersman Company. Of these discharges all except 33 kg N/day and 1.7 kg P/day were diverted by 1975.

The total average pre-diversion phosphorus load can be calculated by adding the following components: 1) upstream non-point Black and Little Black Creek loads, 2) Black Creek downstream non-point loads, 3) direct discharges, and 4) Little Black Creek storm runoff loads (25% of the total storm runoff is already included in the Black Creek downstream loading). The resultant total phosphorus load is 46 kg P/day. The waste diversion should reduce the phosphorus load by approximately 60%. A comparison of the 1975 Mona Lake phosphorus loads to the calculated 1972 pre-diversion load shows a reduction of 65%. Using this same methodology to calculate the post-diversion load the phosphorus load is calculated to be 19 kgms P/day. This estimate includes storm runoff, non-point sources, spray drainage, and undiverted point sources. This calculated load is similar to the 1974 and 1975 measured loads.

Based on the former calculations, future abatement of urban and storm runoff and waste lagoon seepage could reduce the phosphorus load by an additional 50%. Future diversion of the remaining point sources would reduce phosphorus loads by an additional 10%.

Calculations of the pre-diversion dissolved inorganic nitrogen load were conducted using the same techniques. However, only data on total nitrogen were available for storm runoff (Weaver, 1976). As an approximation, the estimated total storm runoff load of 11.7 thousand kg N/year was multiplied to 0.75 to estimate the total dissolved inorganic load. The 1972 pre-diversion load was calculated to be 365 kg N/day. On this basis the diversion project should have reduced the nitrogen load by 55%. Abatement of storm and urban runoff and waste lagoon seepage could reduce the nitrogen load by an additional 50 to 60%. Abatement of the remaining point sources would reduce the nitrogen loads by an additional 15 to 20%.

A comparison of the calculated pre-diversion dissolved inorganic nitrogen load to the 1975 measured load shows a reduction of only 10%. A portion of the reduction was negated by the 50 kg N/day load from spray-site drainage (see Appendix E). Without this the reduction would have been 25%. An additional portion of the reduction was masked by varying non-point loads.

The post-diversion dissolved inorganic nitrogen load was calculated using the above load component analysis; the result was 167 kg N/day. The measured 1975 load was 337 kg N/day. A large portion of this discrepancy results from an underestimation of the non-point nitrogen load to Little Black Creek. The estimated load was 22 kg N/day; the calculated load obtained by subtracting undiverted point loadings from the measured Little Black Creek load was 149 kg N/day. The difference represents unidentified point or non-point sources. Considering this measured but unidentified load the 1972 pre-diversion load is calculated to be 492 kg N/day; the potential reduction in nitrogen load as a consequence of the diversion is then 43%. The observed reduction between this load and the 1975 measured load is 32%.

Summarized in Table 31 are the calculated annual average nutrient loads to Mona Lake for 1972 to 1975 based on observed water quality monitoring data, calculated river flows, wastewater discharge data, and the previous analysis of non-point loading which was used to calculate the 1972 loads because 1972 data on Little Black Creek were limited. The data in Table 31 depicts a 63% reduction in phosphorus loading and a 32% to 45% (1975 vs. 1974) reduction in nitrogen loading. The exact percentages of nutrient load reductions are difficult to calculate for Mona Lake because of uncertainties in non-point loading and the applied river flow correlations.

Spray-irrigation drainage had small effects on loads to Mona Lake. The average load expected to originate from this drainage can be calculated based on Muskegon County measurements of drainage water quality and estimates of the drainage volume (see Appendix E). For 1974 and 1975 these calculations suggest a total phosphorus load addition of 0.6 kg P/day and 45.2 kg N/day of dissolved inorganic nitrogen. The phosphorus load increase was not observed in any of the yearly comparisons of Black Creek data; in fact, a reduction was observed. Part of the anticipated increase in nitrogen load was observed although part of this increase could have been unrelated to the spray-irrigation program.

A simple and approximate evaluation of the reduction in nutrient input to Mona Lake can be achieved by calculating the weighted average tributary concentration resulting from each yearly load. The calculated concentrations are presented in Table 32. As can be seen from these data, a significant reduction in lake phosphorus concentrations are expected. Little can be said regarding nitrogen without information on total nitrogen.

Further evaluation of the reduction in nutrient loads to Mona Lake can be conducted using the Vollenweider model (Vollenweider, 1975). According to this model the minimum eutrophic or dangerous load rate is 0.92 g P/m^2/year while the maximum permissible or oligotrophic loading rate is 0.46 g P/m^2/year. The pre-diversion 1972 phosphorus load to Mona Lake was approximately 6.0 g P/m^2/year. It is apparent that the system is still being loaded with phosphorus at a rate one order of magnitude greater than the maximum loading rate calculated to be necessary to achieve oligotrophic conditions. According to this model the system will maintain in its eutrophic condition despite the diversion of wastes. A comparison of the Mona Lake load to loads for other lakes considered eutrophic reveals the pre- and post-diversion Mona Lake rate to be equal to or larger than those for the other lakes.

Table 31. ESTIMATED TOTAL YEARLY NUTRIENT LOADS TO MONA LAKE, 1972-1975 IN THOUSAND KILOGRAMS PER YEAR

	Total Phosphorus				Dissolved Inorganic Nitrogen			
Year	Municipal & Industrial	Non-Point White River	Spray-Site Drainage	Total	Municipal & Industrial	Non-Point White River	Spray-Site Drainage	Total
1972	10.81 (64.7%)	5.9 (33.3%)	0.0 (0%)	16.71	83.21 (46.3%)	96.37 (53.7%)	0.0	179.58
1973	5.45 (36.4%)	9.35 (62.5%)	0.16 (1.1%)	14.96	42.14 (45.8%)	47.15 (51.3%)	2.69 (2.9%)	91.98
1974	1.06 (13.8%)	6.36 (82.6%)	0.28 (3.6%)	7.70	12.73 (12.8%)	72.29 (72.5%)	14.62 (14.7%)	99.64
1975	0.63 (10.6%)	4.89 (82.7%)	0.39 (6.6%)	5.91	12.32 (10.0%)	91.54 (79.4%)	19.14 (15.6%)	123.00

Table 32. FLOW WEIGHTED AVERAGE CONCENTRATIONS OF TOTAL PHOSPHORUS AND DISSOLVED INORGANIC NITROGEN IN TRIBUTARIES TO MONA LAKE

Year	Total Phosphorus (μg P/ℓ)	Dissolved Inorganic Nitrogen (μg N/ℓ)
1972	379	3903
1973	203	1320
1974	104	1353
1975	83	1820

Table 33 is a total nutrient budget for Mona Lake. The most notable observation from these calculations is the negative retention observed for phosphorus in 1975. This suggests a response of the lake attempting to buffer the effect of the nutrient reduction. It is possibly a result of sediment nutrient release. This is occurring because of the reduction in the concentration of phosphorus in the water, which in turn disturbed the former sediment-water nutrient balance.

Summary -- Black Creek is the major hydrologic contributor to Mona Lake. Little Black Creek was the most significant nutrient source to Mona Lake prior to diversion and during the post-diversion period was equally as important as Black Creek. Very high nutrient concentrations were observed in Little Black Creek which originated from industrial and municipal discharges and urban runoff. Significant reductions in phosphorus loads to Mona Lake were observed during the investigation. These were a consequence of the diversion of the Muskegon Heights Wastewater Treatment Plant discharge and improved treatment by the Kersman Company. The approximate 65% reduction in phosphorus observed was similar to that predicted. For dissolved inorganic nitrogen a 40 to 55% reduction was expected and the observed reduction was calculated to be approximately 30 to 45%. The Vollenweider model predicts no change in the trophic status of Mona Lake.

LAKE-RELATED CONSIDERATIONS

It is important to note at the onset of any discussion of Mona Lake that the physical, chemical, and biological parameters measured are generally many times greater than those observed in either White or Muskegon Lakes which are comparatively much less productive. Additionally, the dominant species of phytoplankton differ from those observed in Muskegon and White Lakes: non-nitrogen fixing blue-green algae and centric diatoms for Mona Lake and nitrogen-fixing blue-greens and pennate diatoms for White and Muskegon Lakes (USEPA 1975 a,c,d). This is, certainly, a result of chemical differences among the lakes.

Spatial and Seasonal Distributions -- To evaluate horizontal distributions in Mona Lake, the concentrations of total dissolved phosphorus and total inorganic nitrogen were observed at Stations 3 and 4. The former station is located at the east end of the lake near the mouth of Black Creek (a major tributary to Mona Lake) while the latter station is at the west end and reflects average lake conditions.

Total inorganic nitrogen values at Stations 3 and 4 fluctuated, but neither station consistently recorded higher or lower values than the other (see Figure 92). In the case of total dissolved phosphorus, surface values at the two stations were quite similar (see Figure 93). Occasional peaks were observed during the summer and fall in the surface waters of Station 4; probably due to the influence of Little Black Creek, a major source of nutrients.

Table 33. A SUMMARY OF THE YEARLY AVERAGE FLUX OF NUTRIENTS LEAVING AND ENTERING MONA LAKE IN KILOGRAMS PER DAY

	Year	Total Dissolved Phosphorus as P	Total Phosphorus as P	Nitrate + Ammonia as N	Silicon as Si	Chloride as Cl
Entering	1973	27.1	38.8	252	445	5674
	1974	11.1	21.1	273	579	7379
	1975	7.0	16.2	333	721	8361
Leaving	1973	16.2	26.7	74	69	4401
	1974	3.8	13.5	114	161	5802
	1975	17.2	20.3	141	438	8385
Percent Retained in Mona Lake	1973	41%	31%	71%	85%	23%
	1974	66%	64%	58%	63%	21%
	1975	-140%	-25%	58%	39%	0%

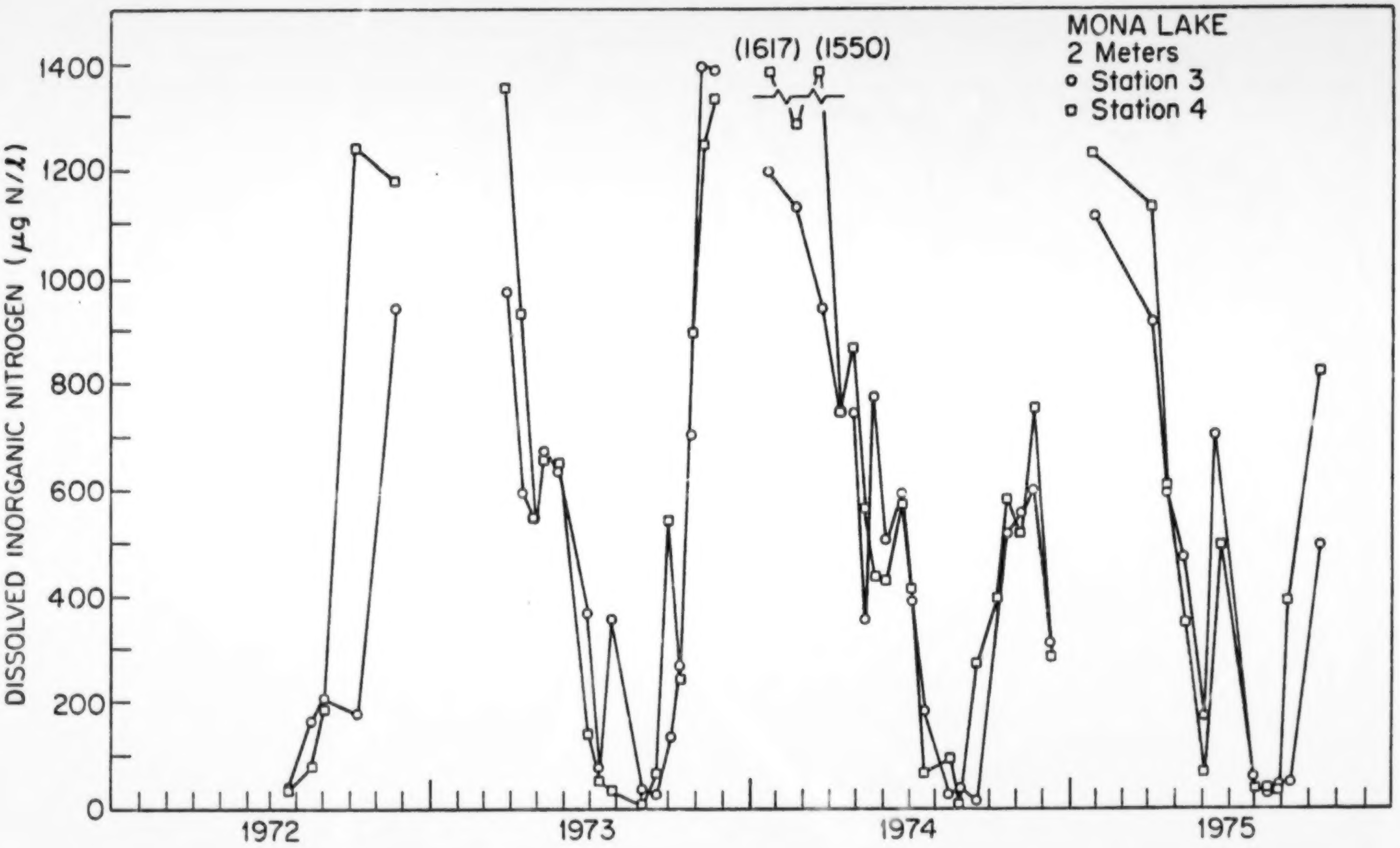


Figure 92. Dissolved inorganic nitrogen concentrations at Stations 3 and 4 in Mona Lake, 1972-1975.

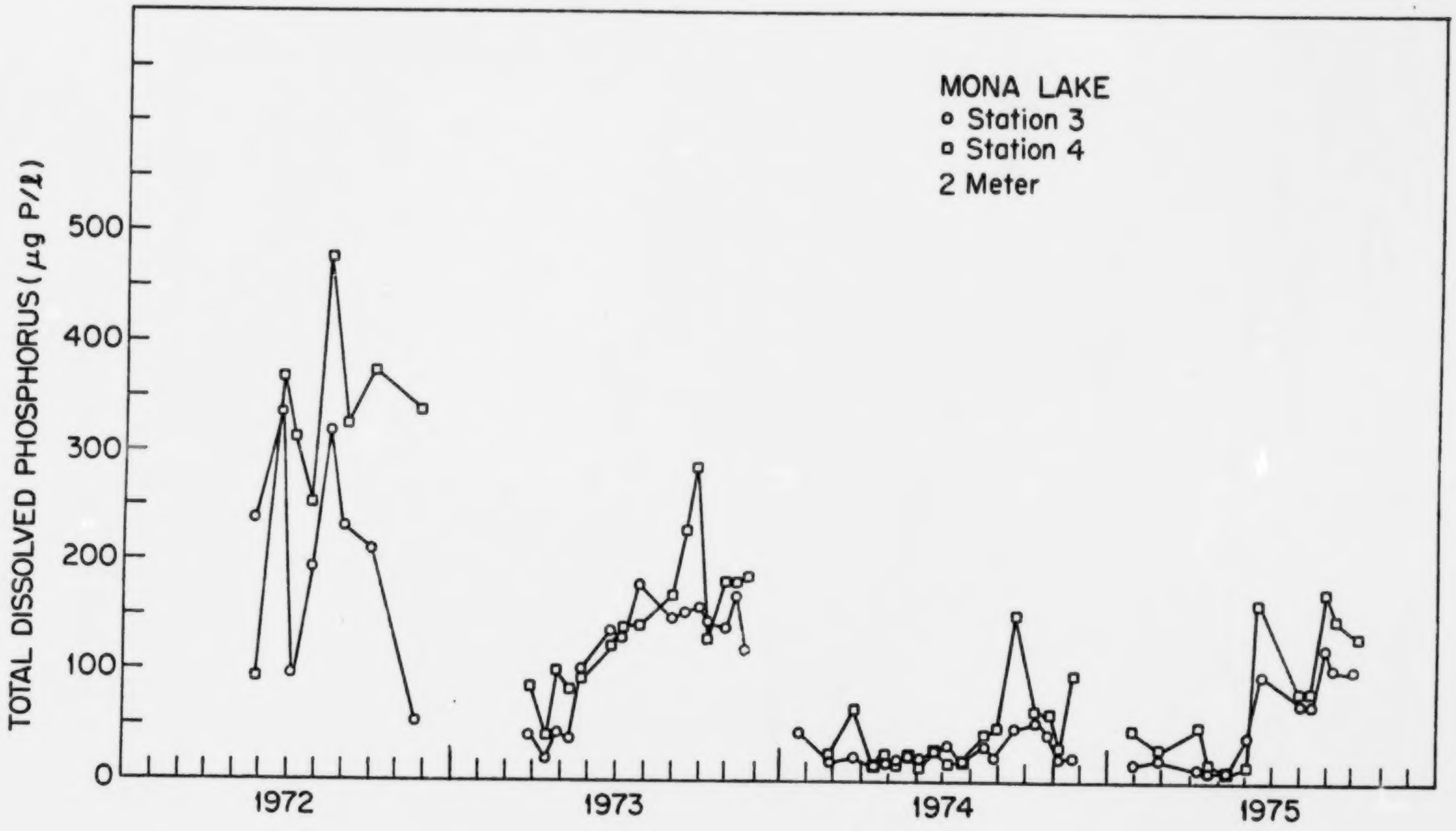


Figure 93. Total dissolved phosphorus concentrations at Stations 3 and 4 in Mona Lake, 1972-1975.

Vertical and seasonal distribution of chemical and biological parameters in Mona Lake follows to a large extent many of the classic patterns observed in White and Muskegon Lakes. Surface nitrate levels peak in March (exceeding 1,000 μg N/ℓ) and then rapidly decline with increasing phytoplankton production in the spring and early summer. By July (August in 1974) the nitrate has been depleted (to the limit of detection) and remains low into the fall. There is no replenishment of surface nitrate through the summer months (see Figure 94).

Surface water ammonia concentrations vary somewhat, but are generally quite low compared to bottom water levels (see Figure 95). Spring and fall peaks in ammonia concentration respond to peaks in loading and mixing with bottom waters high in ammonia. Accumulation of ammonia in the surface waters is observed during the period October through February. This is the result of the introduction of ammonia from the bottom waters during a period of low phytoplankton activity. Nitrification and phytoplankton assimilation quickly reduces these levels to concentrations approaching the limit of detection. Although nitrate concentrations are reduced to zero in the surface waters in the summer, this is not the case for ammonia. The minimum reported summer surface ammonia concentrations were: 28.5 μg N/ℓ in 1972, 8.3 μg N/ℓ in 1973, 9.0 μg N/ℓ in 1974, and 16.7 μg N/ℓ in 1975. The average summer concentrations in the surface waters were 2-10 times this value. The surface summer minimum and surface summer average for ammonia in Mona Lake was twice the value reported for either White or Muskegon Lakes. The lack of complete depletion of ammonia in the surface waters in the summer may account for the absence of the nitrogen-fixing blue-green algae as observed in Muskegon and White Lakes (USEPA, 1975d).

The hypolimnetic waters of Mona Lake become anoxic rapidly during the spring and remain totally anoxic throughout the summer (see Figure 96). The occurrence of denitrification of nitrate to nitrogen gas is considered a possibility. This is more the case in Mona Lake than in White or Muskegon Lakes because of the more severe conditions of hypolimnetic anoxia. Nitrate concentrations during the spring, late fall, and winter in the bottom waters are high and are generally comparable to surface values. Ammonification on the other hand may be an important process in the bottom waters of Mona Lake especially during the summer months. Extremely high levels of ammonia-nitrogen were reported in the bottom waters during 1972 (8,536 μg N/ℓ) and 1973 (9,484 μg N/ℓ). Significantly smaller peaks were reported during 1974 (1,907 μg N/ℓ) and 1975 (2,109 μg N/ℓ). Bottom water concentrations of ammonia fall rapidly in October (turnover) and remain low until anoxia and ammonia production begin again in June.

Three forms of phosphorus were measured in Mona Lake: soluble reactive, total dissolved, and total. Surface water concentrations of both soluble reactive and total dissolved phosphorus followed a similar pattern. In Mona Lake, as opposed to Muskegon and White Lakes, the soluble reactive phosphorus is the major component of the dissolved fraction. This may indicate a rapid turnover or mineralization of the organic fraction,

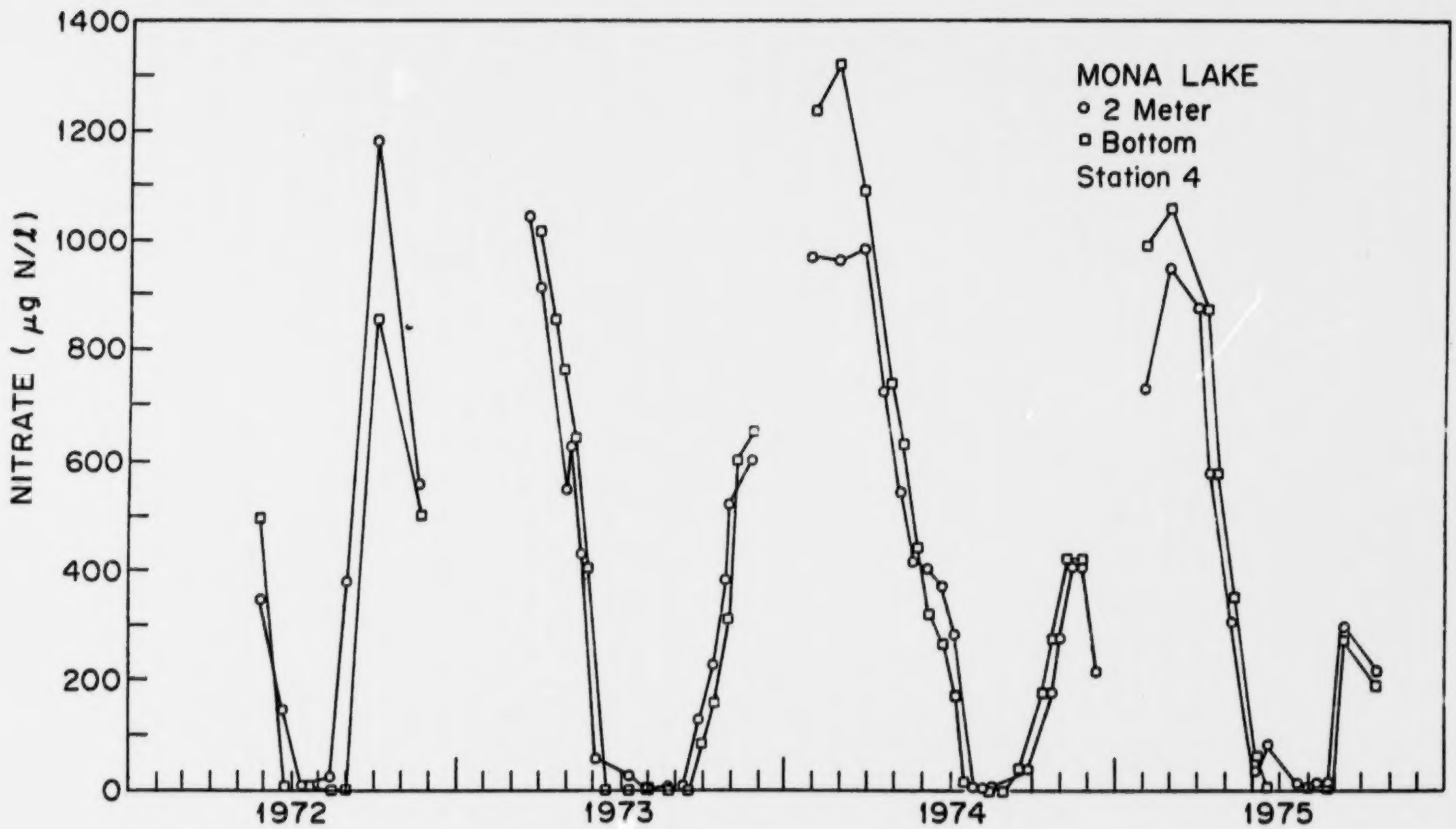


Figure 94. Nitrate concentrations in Mona Lake, 1972-1975.

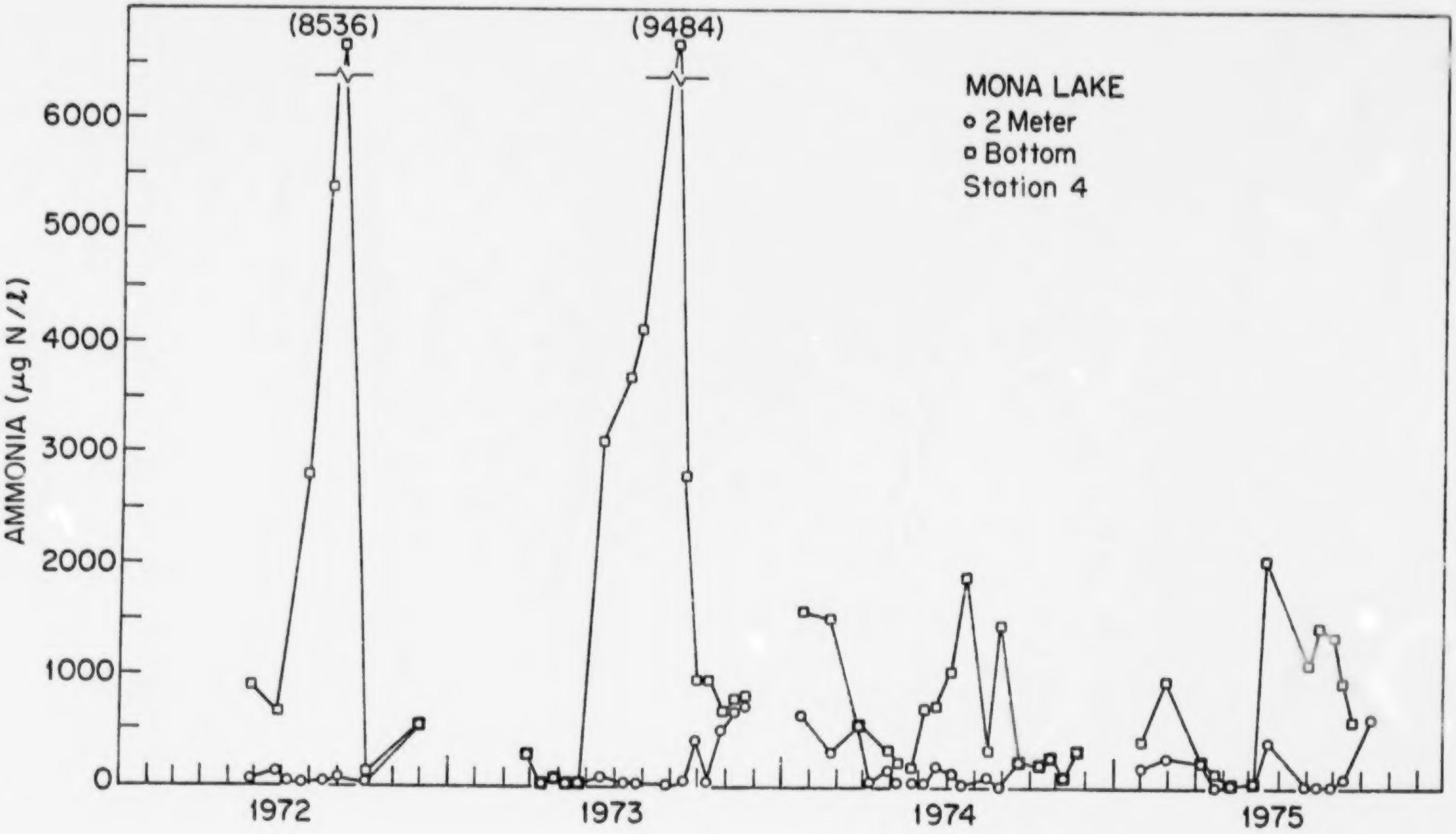


Figure 95. Ammonia concentrations in Mona Lake, 1972-1975.

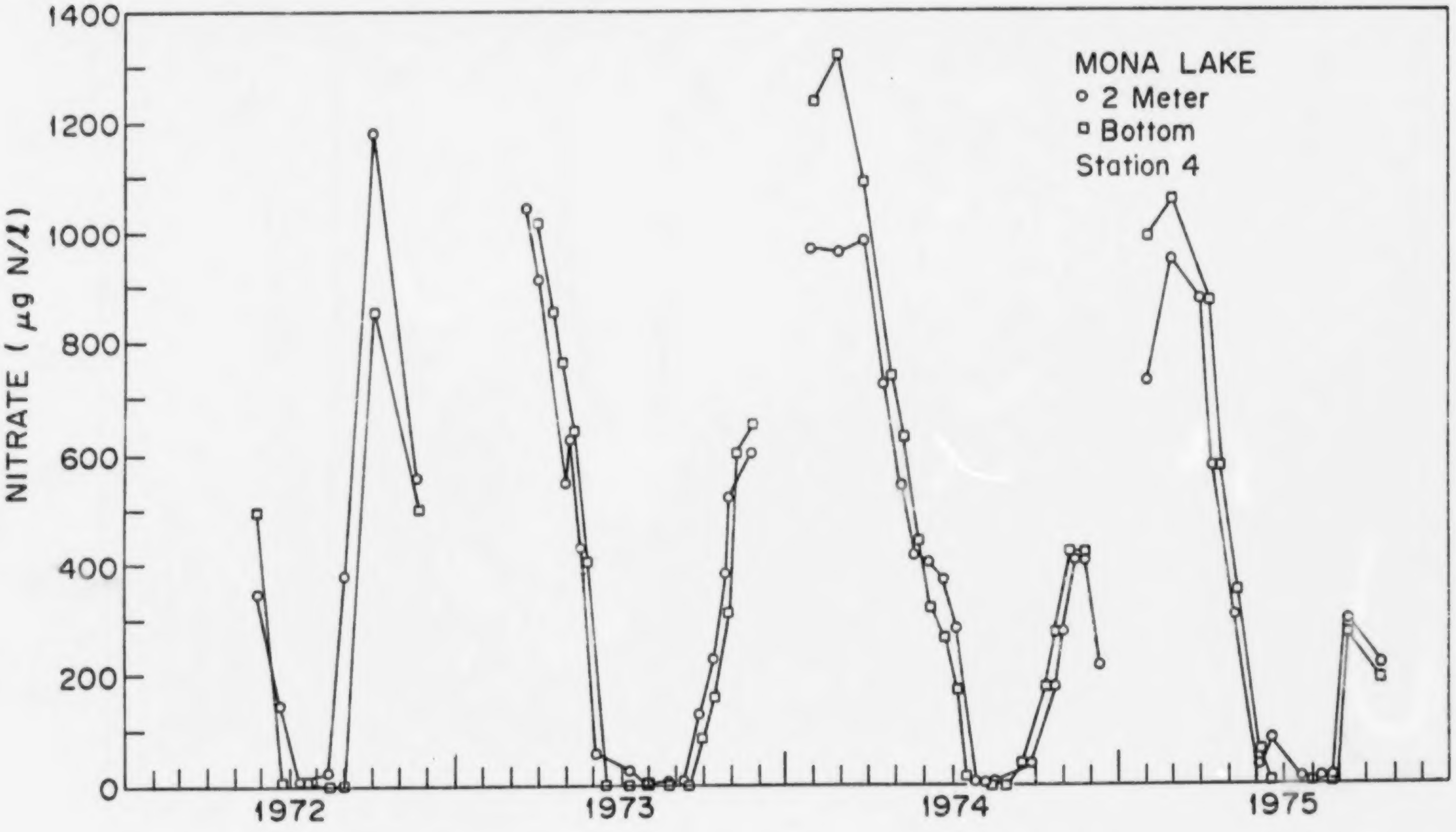


Figure 94. Nitrate concentrations in Mona Lake, 1972-1975.

transfer of soluble reactive phosphorus from the bottom waters or an amount of phosphorus present in excess of that which can be used by the phytoplankton. Surface water soluble reactive and total dissolved phosphorus concentrations are generally highest during the late fall (100 - 400 μg P/ℓ), after replenishment from the hypolimnetic waters. Concentration of phosphorus then decrease slowly until they are depleted in the summer (see Figures 97 and 98). The severity of depletion has increased from 1972-1975, but sufficient phosphorus remains to support large populations of phytoplankton. The increased depletion is probably a result of less total available phosphorus not necessarily greater algal growth.

Peak total phosphorus concentrations in the surface waters of Mona Lake occurred at approximately the same time as peaks in primary productivity (see Figure 99). The total phosphorus is, however, probably responding to load variation and sediment release. Dissolved phosphorus concentrations in Mona Lake are much in excess of what can be assimilated by the phytoplankton and the particulate fraction therefore accounts for much less than half of the total phosphorus. It would be expected that the phytoplankton would convert the dissolved phosphorus to the particulate form if this were not the case.

The dynamics of the phosphorus cycle in the bottom waters are dominated, as in the other lakes, by soluble reactive phosphorus-iron interactions. Except during periods of anoxia, bottom water soluble reactive phosphorus concentrations follow those of the surface waters. During the summer months when no oxygen exists in the hypolimnion, tremendous quantities of phosphorus and dissolved iron are released from the sediment according to the mechanism discussed earlier (see Figures 97 and 100). Soluble reactive phosphorus concentrations approaching or exceeding 1 mg P/ℓ were reported on numerous occasions during the summer. Such large amounts of organic matter are generated in Mona Lake that hypolimnetic oxygen depletion can occur, even in the winter months. Although complete deoxygenation was not observed (1 - 5 mg/ℓ remains), conditions could have been met in the sediments for the release of phosphorus and iron to the bottom waters.

Silicon concentrations in Mona Lake follow a pattern of depletion associated with spring phytoplankton uptake. Silicon concentrations are reduced to zero in Mona Lake as early as April and May. Replenishment (through tributary loads) is very rapid and begins in June. A second period of depletion begins in September which is equivalent to the spring depletion period. Concentrations then increase to the winter maximum (see Figure 101).

The silicon cycle is in marked contrast to that observed in either White Lake or Muskegon Lake. For these lakes there was only one, much less rapid silicon uptake period which did not result in concentrations approaching zero. A slight replenishment occurred in mid-summer followed by

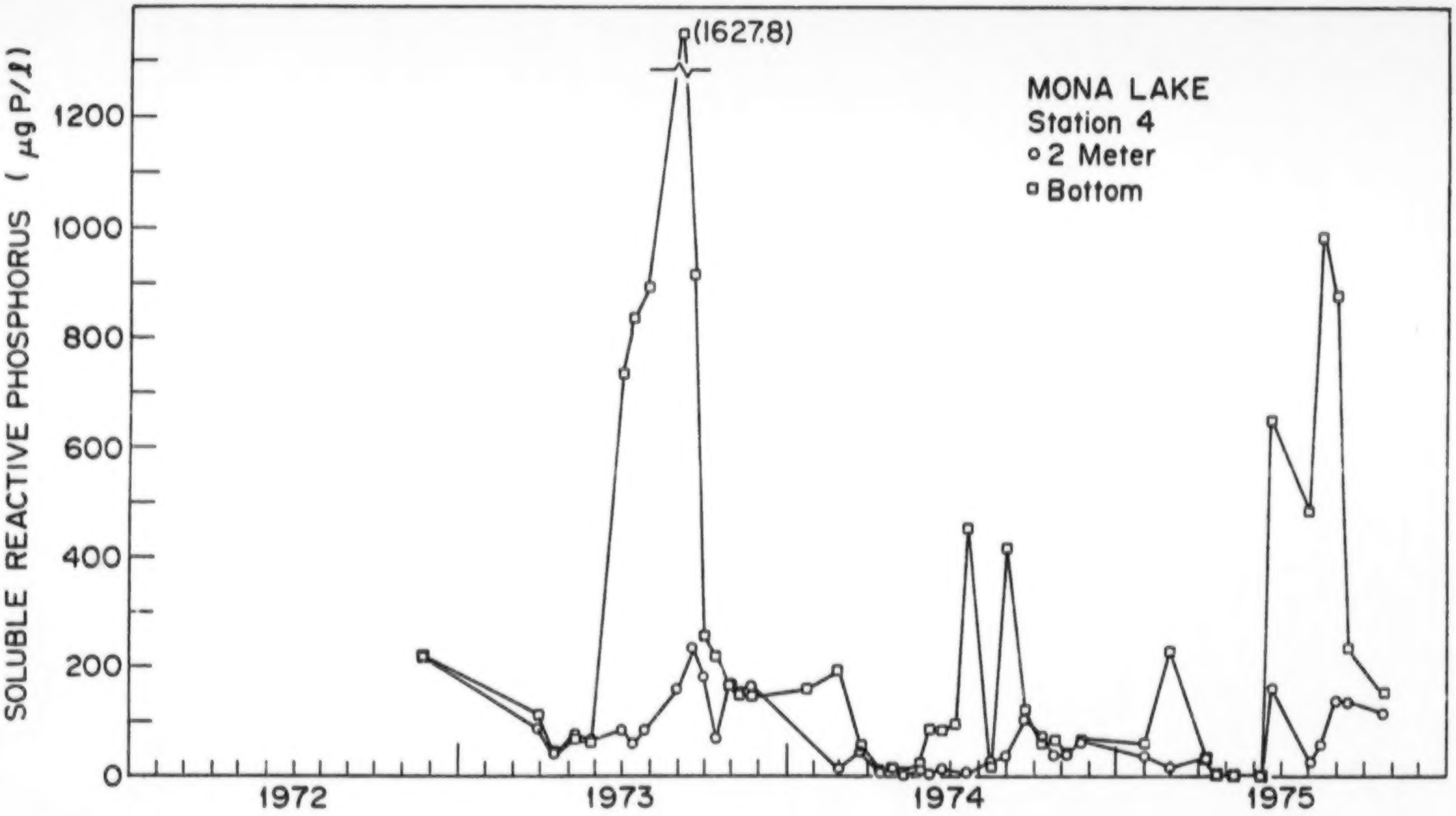


Figure 97. Soluble reactive phosphorus concentrations in Mona Lake, 1972-1975.

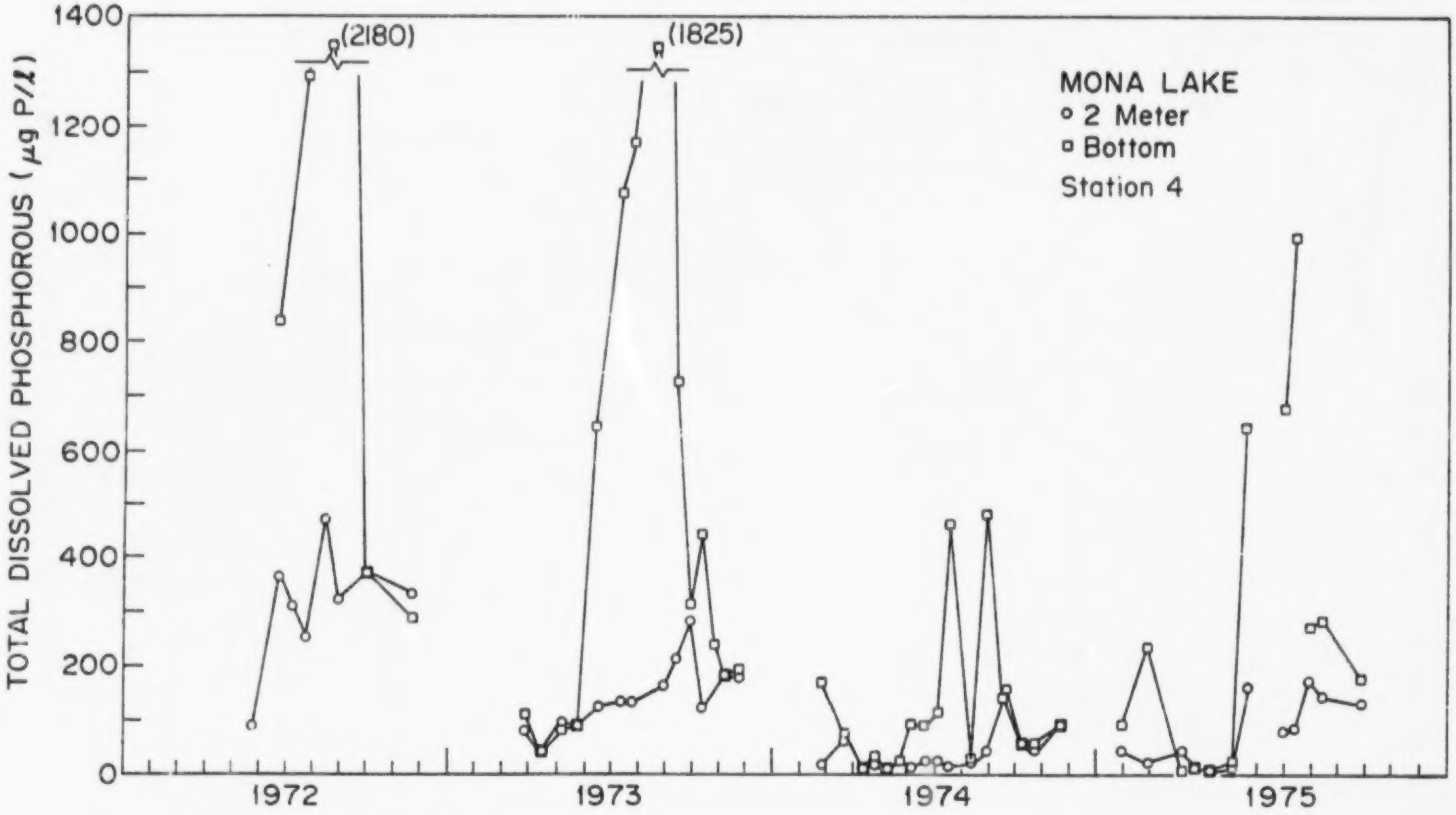


Figure 98. Total dissolved phosphorus concentrations in Mona Lake, 1972-1975.

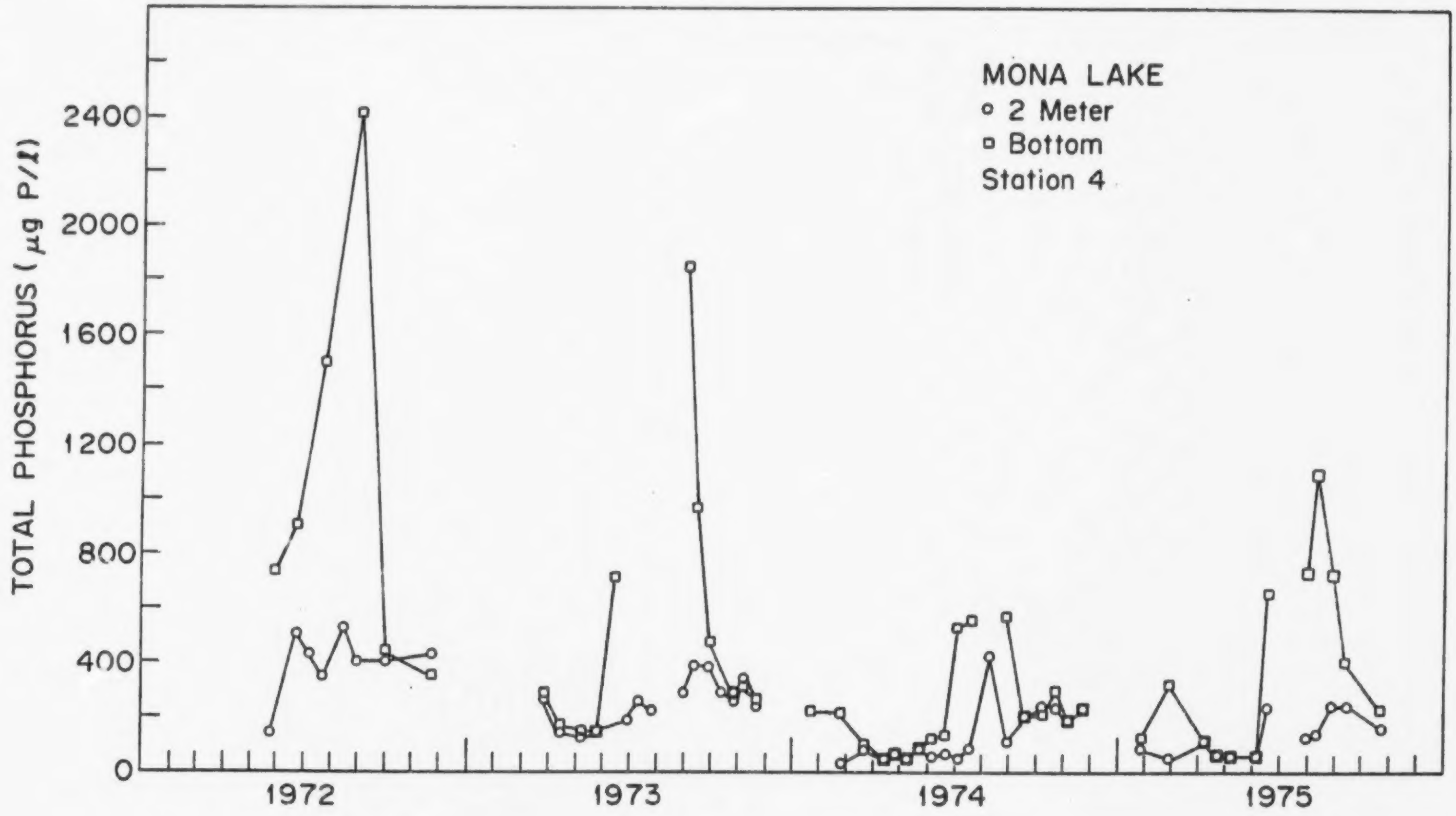


Figure 99. Total phosphorus concentrations in Mona Lake, 1972-1975.

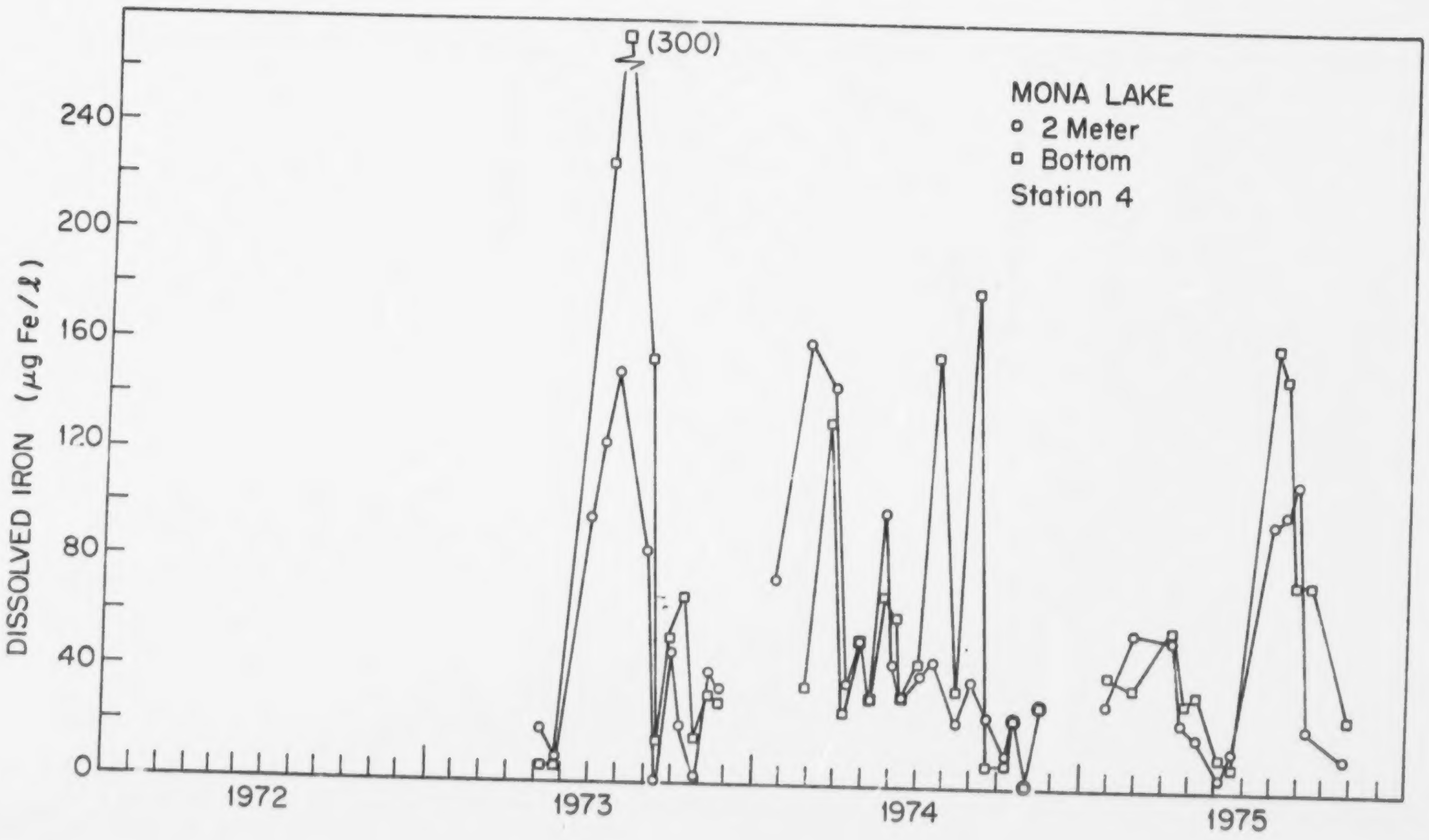


Figure 100. Dissolved iron concentrations in Mona Lake, 1973-1975.

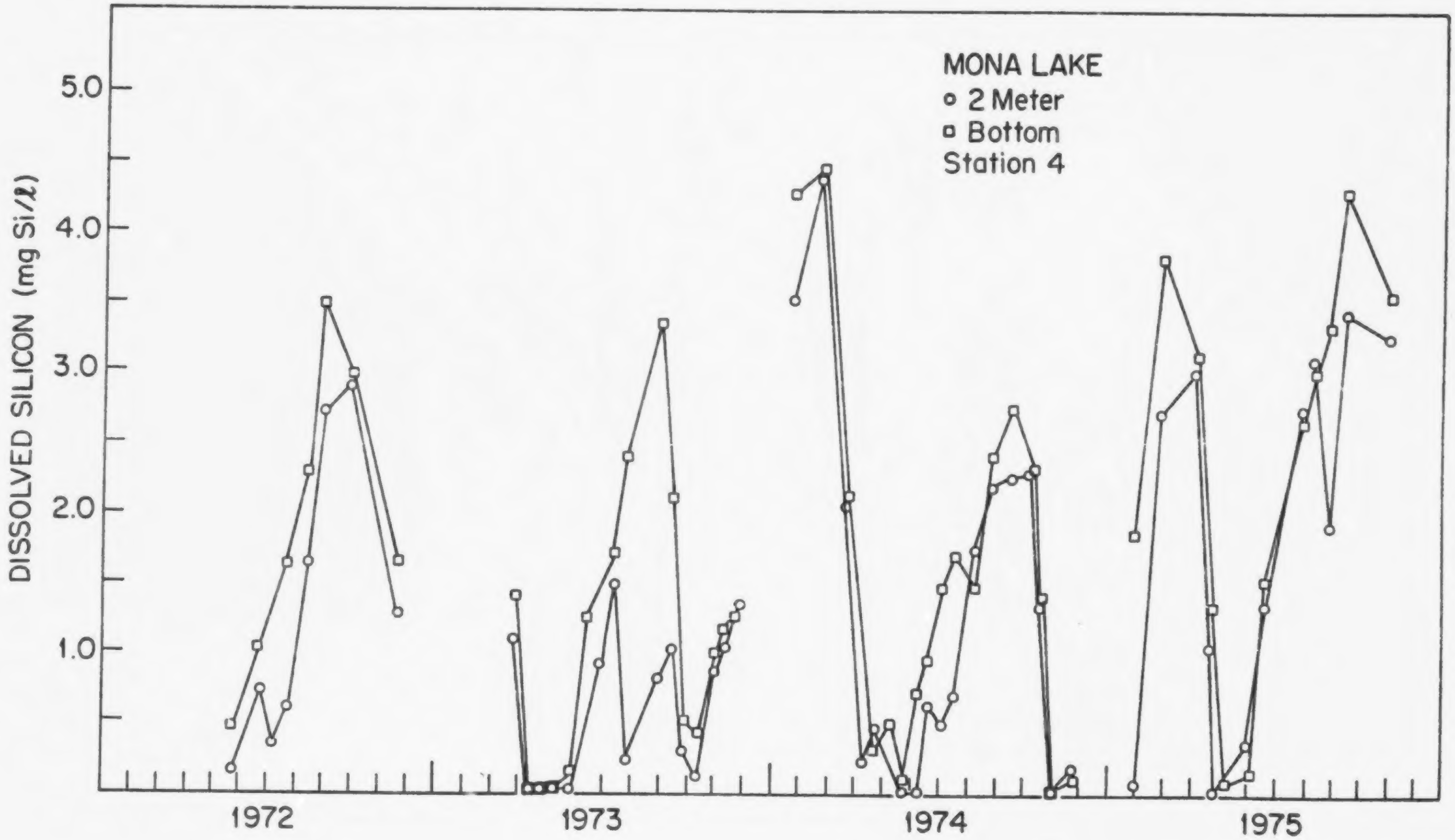


Figure 101. Dissolved silicon concentrations in Mona Lake, 1972-1975.

a fall uptake. The comparative magnitude of the dynamics of the two less productive lakes and Mona Lake is notable and reflects the higher productivity in Mona Lake.

Chlorophyll *a* and primary productivity levels are elevated for a longer part of the year in Mona Lake than in either White or Muskegon Lakes; average levels in Mona Lake are also much higher (see Figures 102 and 103). Only partial data are available for 1972 and 1975 in the mid-summer period. From data gathered in 1973 and 1974, however, it is apparent that the only time when chlorophyll *a* values are not high is in the winter months from January through March. Peak chlorophyll *a* values generally occur during the month of October. Primary productivity values follow a similar trend except that peak values generally occur in August. Interpretation of trends in chlorophyll *a* concentration is complicated by the fact that algicides were applied to Mona Lake during the summers of 1970 and 1975. The algicide, month of application, and quantity applied are presented in Table 34.

The application of algicides has had two effects on phytoplankton levels in Mona Lake. First, the populations during the peak summer recreation months have been controlled. Second, the peak phytoplankton bloom has been "postponed" until October or November. The use of algicides may make the average chlorphyll *a* levels appear lower than they would be without algicide and thus obscure the influence of the diversion project.

Nutrient ratios were calculated for Mona Lake as for Muskegon and White Lakes. The winter N/P ratio in the surface waters was 39-47:1 in 1974 and 1975. The bottom water winter N/P ratios were 16-26:1 for this same period. These ratios indicate phosphorus limitation. The summer surface N/P ratios in Mona Lake were <1:1 for all years sampled (1972-1975) except 1974 when it was 4:1; these ratios are indicative of nitrogen limited conditions. The low N/P ratios result from very high phosphorus levels; ammonia nitrogen does not become completely depleted in the surface waters. The ratio of winter dissolved inorganic nitrogen to summer total dissolved phosphorus in the bottom waters was 14:1 in 1974 and 4:1 in 1975. These values indicate nitrogen limitation. Calculations of N/P ratios in the Mona Lake loads demonstrate generally high nitrogen levels. The ratios were 9:1, 24:1, and 42:1 for 1973, 1974, and 1975. Insufficient data were available to calculate a 1972 ratio. The higher ratios in 1974 and 1975 indicate the phosphorus diversion. It is, however, perhaps useless to calculate any of these ratios for Mona Lake due to the extremely high nitrogen and phosphorus levels. Growth in Mona Lake was likely limited by other environmental factors (light, time, etc.) or the algicide application.

Long-Term Changes -- In the period since diversion there has been a significant change in the phytoplankton nutrient concentrations in Mona Lake. Calculations of changes in various parameters are presented in Table 35.

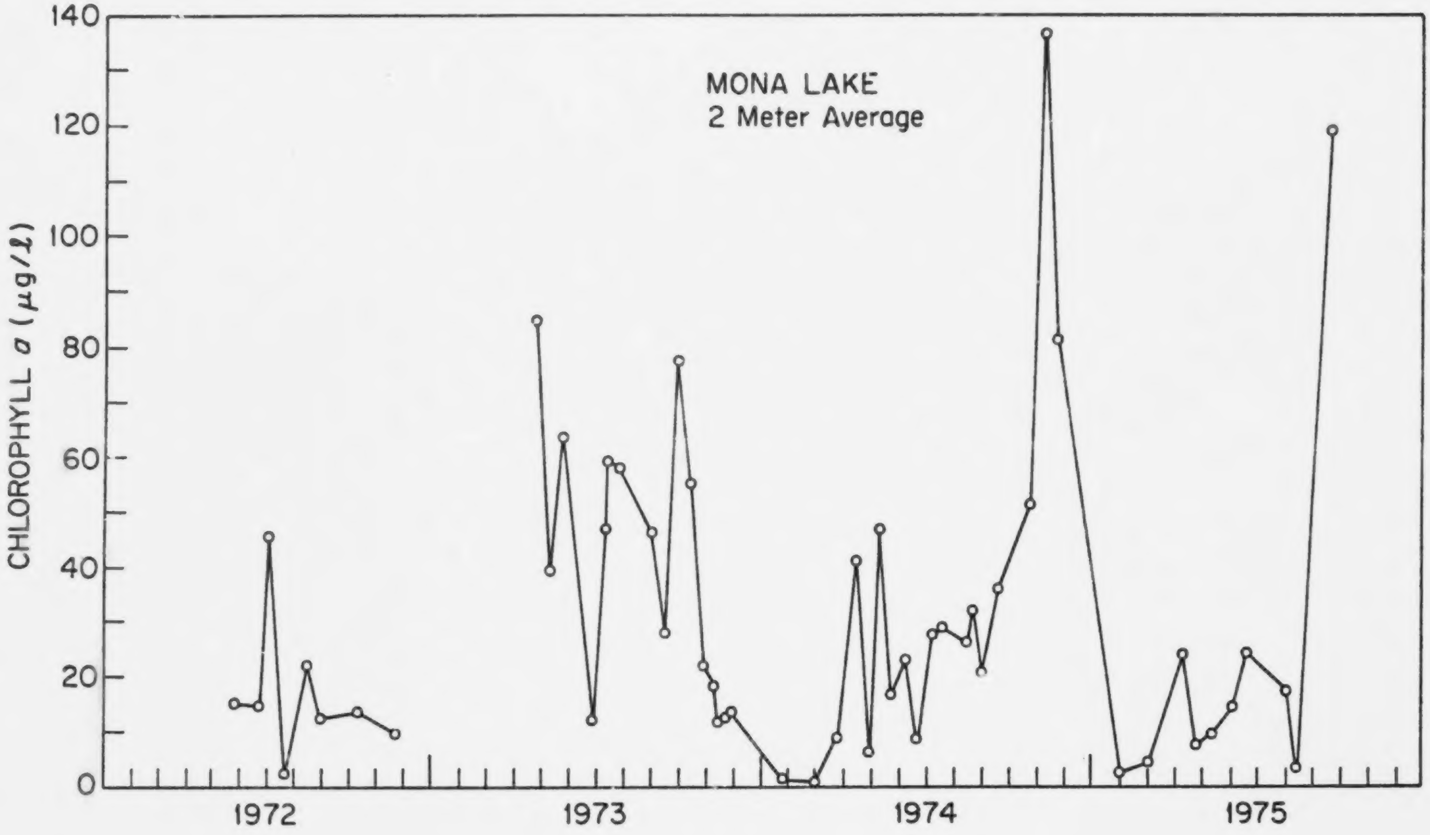


Figure 102. Chlorophyll *a* concentrations in Mona Lake, 1972-1975.

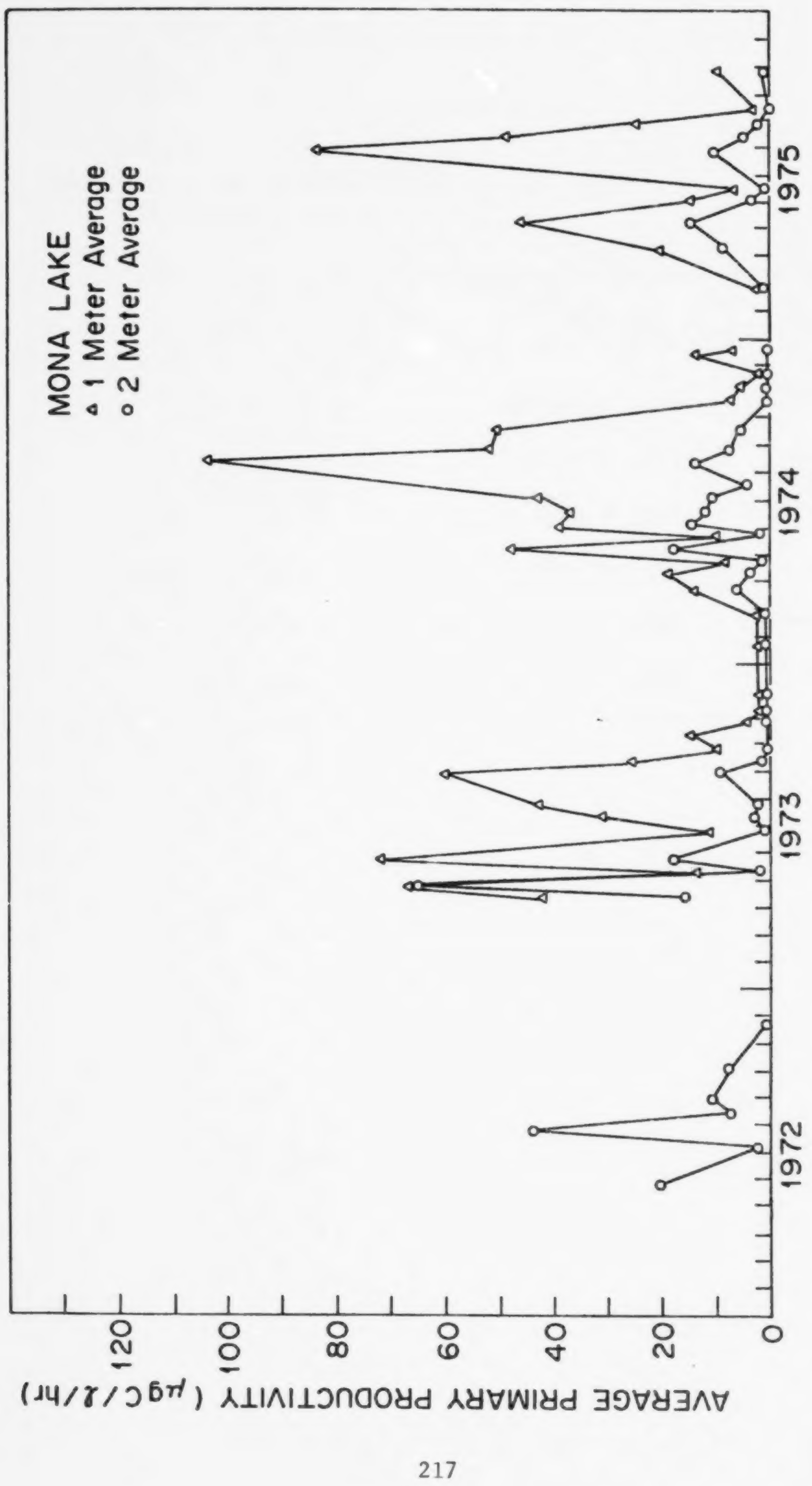


Figure 103. Primary productivity rates in Mona Lake, 1972-1975.

Table 34. ALGICIDE APPLICATION TO MONA LAKE
(Michigan DNR, 1976)

Year	Month	Algicide	Quantity
1970	June & August	Copper Sulfate	4200 pounds
1971	June & August	Copper Sulfate	4200 pounds
1972	June & August	Copper Sulfate	4200 pounds
1973	June & August	Copper Sulfate	4200 pounds
1974	June, July, & August	Endothall	75 gallons
1975	June, July, & August	Endothall	300 gallons

Table 35. AVERAGE ANNUAL VALUES FOR SELECTED WATER QUALITY PARAMETERS IN MONA LAKE, 1972-1975

Year	Ammonia (μg N/ℓ)	Nitrate (μg N/ℓ)	Dissolved Inorganic Nitrogen (μg N/ℓ)	Total Dissolved Phosphorus (μg P/ℓ)	Total Phosphorus (μg P/ℓ)	Soluble Reactive Phosphorus (μg P/ℓ)	Chlorophyll *a* (μg/ℓ)	Secchi Disc (m)
				Surface				
1972	126	321	447	261	338	86	17.6	1.21
1973	183	367	550	128	226	95	40.0	0.92
1974	160	417	577	34	108	25	34.4	1.05
1975	156	337	493	68	134	49	29.8	0.92
				Bottom				
1972	1374	321	1695	661	675	116		
1973	1199	362	1561	344	380	302		
1974	389	451	840	72	158	64		
1975	476	353	829	194	259	172		

Although nitrate concentrations and surface ammonia concentrations have remained relatively constant, a 65% reduction in bottom water ammonia concentrations has been observed. More importantly, similar and more striking reductions in total dissolved and total phosphorus were noted. Levels of total dissolved phosphorus in both surface and bottom waters decreased by more than 70% while concentrations of total phosphorus were reduced by more than 60%. This change has not had a measured effect on primary productivity. Average chlorophyll *a* levels in 1973-1975 were markedly higher than in 1972. There is a slight yearly reduction in chlorophyll *a* levels between 1973 and 1974 and again between 1974 and 1975. This reduction may be a result of the diversion project, but this is unlikely due to the tremendous amounts of nutrients remaining. It is more likely that variation in average chlorophyll *a* levels between 1972 and 1975 was a result of algicide applications and changes in sampling frequency. Because of the algicide application, all analysis regarding primary productivity and chlorophyll *a* is complicated. Seasonal cycles were distorted and the maximum phytoplankton growth occurred late in the fall. The effect of the diversion on algal growth cannot be easily discerned.

As discussed earlier, Dillon and Rigler (1974, 1975) have related water clarity (Secchi disc), chlorophyll *a*, and total phosphorus (see Figures 76 and 77). Because of the particularly high nutrient levels in Mona Lake and the application of algicides, these relationships cannot be directly applied to the Mona Lake data. Summer chlorophyll *a* levels predicted using these correlations were twice or more than those observed. Based strictly on these concentrations, 1975 phosphorus levels would have to be reduced an additional three-quarters or more to gain a one-meter increase in Secchi disc under natural conditions.

The dissolved oxygen resources of Mona Lake may have changed slightly during the period of this study. The duration of anoxia appeared to be slightly less in 1975 than in 1974 and 1973. This could be a result of lessened biological oxygen demand loading and autochthonous production. However, significant amounts of nitrate were not observed in the summer bottom waters. This suggests denitrification or inhibition of nitrification. Both occurrences are related to anoxic conditions. Bottom water ammonia concentrations were less in 1974 and 1975.

Summary -- A calculated 60% reduction in total phosphorus load for Mona Lake was predicted and observed. A 40 to 55% reduction in dissolved inorganic nitrogen load was predicted and an estimated 30 to 45% reduction was observed. The actual nutrient and water quality conditions for 1972 and 1975 are presented in Figure 104 along with data from Lakes Michigan and Erie for comparison. The large reduction (65%) in phosphorus levels in Mona Lake may have resulted in a slight decrease in chlorophyll *a*; this is not clear due to the use of algicides in the lake. Nutrient concentrations remained very high compared to other lakes. Some slight improvements in hypolimnetic anoxia were observed. It may be expected that further improvement in water quality could result from reduced phytoplankton growth leading to shorter periods of hypolimnetic anoxia. This would retard phosphorus release from the sediments and thus further reduce algal growth.

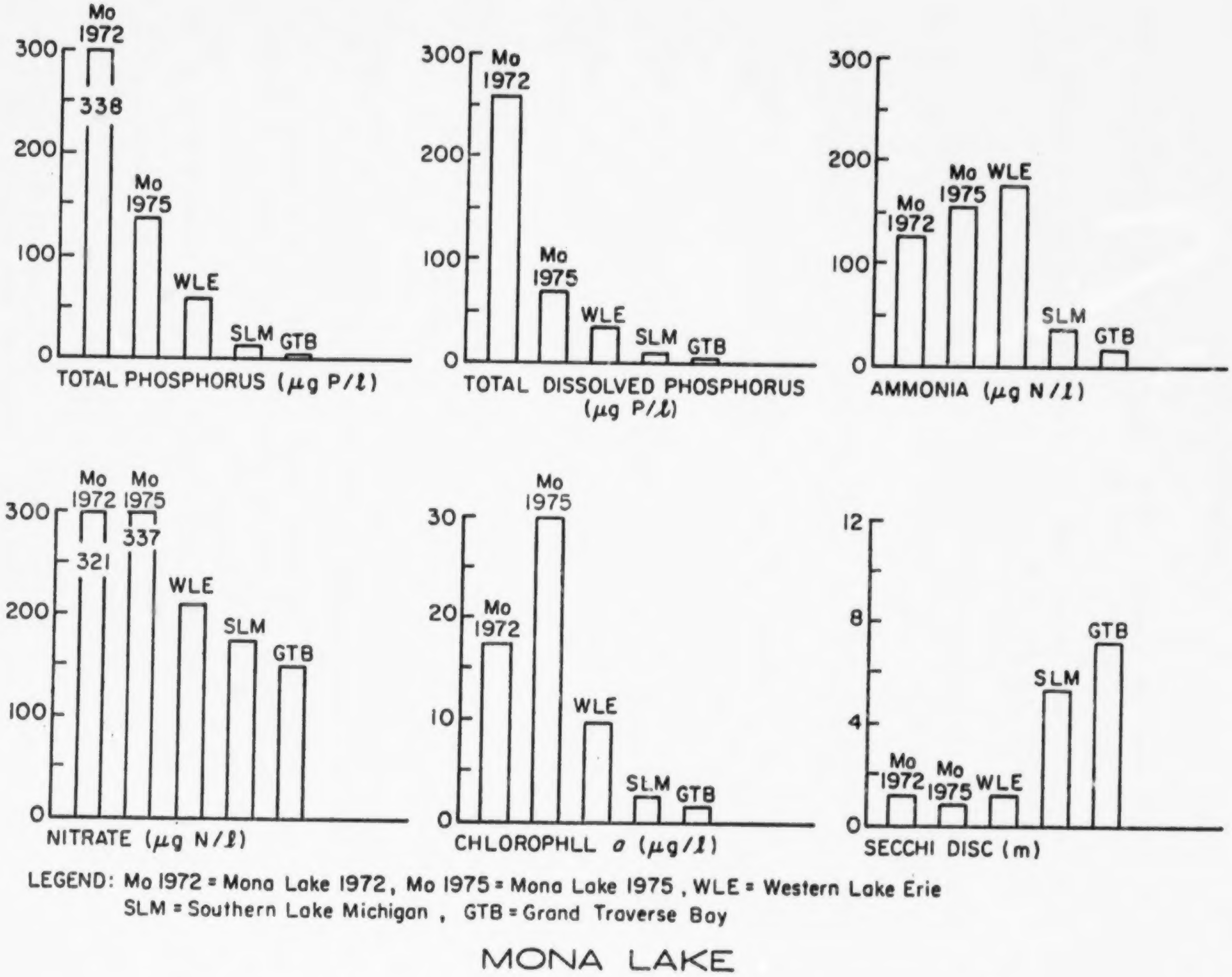


Figure 104. Pre- and post-diversion status of selected two-meter, yearly average water quality parameters in Mona Lake.

LITERATURE CITED

Allen, H.E. and J.R. Kramer. 1972. Nutrients in Natural Waters. Wiley-Interscience, New York. 457 pp.

Anderson, R.R., R.G. Brown, and R.D. Rappleye. 1966. The mineral content of Myriophyllum spicatum L. in relation to its aquatic environment. Ecology, 47 (5):844-846.

APHA, AWWA, and WPCF. 1971. Standard Methods for the Examination of Water and Wastewater. 13th Edition. American Public Health Association, Washington, D.C. 874 pp.

Auer, M.T. and R.P. Canale. 1976. Muskegon Algal Nutrient Bioassay Study—White Lake Project. The University of Michigan, Ann Arbor. 110 pp.

Auer, M.T., R.P. Canale, and P.L. Freedman. 1976. The Limnology of Grand Traverse Bay. Michigan Sea Grant Technical Report No. 47. Ann Arbor, Michigan. 244 pp.

Bauer Engineering, Inc. 1971. The Muskegon County Wastewater Management System. Pamphlet, privately published. Chicago, Illinois. 16 pp.

Byrnes, B.H., D.R. Keeney, and D.A. Graetz. 1972. Release of ammonium-N from sediments to waters. Proc. 15th Conference Great Lakes Res., pp. 249-259.

Caines, L.A. 1965. Phosphorus content of some aquatic macrophytes with special emphasis on seasonal fluctuations and application of phosphorus fertilizers. Hydrobiologia, XXV:289-301.

Chaiken, E.I., S. Poloncsik, and C.D. Wilson. 1973. Muskegon Sprays Sewage Effluents on Land. ASCE-Civil Engineering Reprint. 5 pp.

DeMarte, J.A. and R.T. Hartman. 1974. Studies on absorption of ^{32}P, ^{50}Fe and ^{45}Ca by water-milfoil (Myriophyllum exalbescens). J. Ecology, 55: 189-194.

Denny, P. 1972. Sites of nutrient adsorption in macrophytes. J. Ecology, 60:819-829.

Dillon, P.J. and F.H. Rigler. 1974. The phosphorus-chlorophyll relationship in lakes. Limnol. Oceanogr., 19:767-773.

Dillon, P.J. and F.H. Rigler. 1975. A simple method for predicting capacity of a lake for development based on lake trophic status. J. Fish. Res. Bd. Canada, 32:1519-1531.

Edmondson, W.T. (ed). 1959. Ward and Whipple's Fresh Water Biology. 2nd Edition. John Wiley & Sons, New York. 1248 pp.

Edmondson, W.T. 1972. Nutrients and phytoplankton in Lake Washington. pp. 172-188. In: G.E. Likens (ed), Nutrients and Eutrophication: The Limiting Nutrient Controversy. ASLO, Lawrence, Kansas.

Gerloff, G.C. 1973. Plant analysis for nutrient assay of natural waters. Office of Research and Monitoring, U.S. Environmental Protection Agency, Washington, D.C. 66 pp.

Goldman, C.R. 1972. The role of minor nutrients in limiting the productivity of aquatic ecosystems. pp. 21-33. In: G.E. Likens (ed), Nutrients and Eutrophication: The Limiting Nutrient Controversy. ASLO, Lawrence, Kansas.

Harter, R.D. 1968. Adsorption of phosphorus by lake sediment. Soil Sci. Amer. Proc., 32:514-518.

Johnson, D.M. 1976. Atmospheric Inputs of Trace Metals and Nutrients to Saginaw Bay. The University of Michigan, Ann Arbor (Ph.D. dissertation).

Keachie, P. 1967. Personal communication.

Keenan, J.D. and M.T. Auer. 1974. The influence of phosphorus luxury uptake on algal bioassays. J. Water Poll. Contr. Fed., 46(3):532-542.

Kerr, P.C., D.F. Paris, and D.L. Brockway. 1970. The Interrelation of Carbon and Phosphorus in Regulating Heterotrophic and Autotrophic Populations in Aquatic Ecosystems. Water Poll. Contr. Res. Ser. 16050 FGS 07/70, Fed. Water Qual. Admin., Dept. Interior, Washington, D.C.

Lee, G.F. 1970. Factors Affecting the Transfer of Materials Between Water and Sediment. Eutrophication Information Program. University of Wisconsin, Water Resources Center, Madison. 50 pp (NTIS Pub. No. PB195220).

Meier, P. 1977. Summary of Benthic Macroinvertebrates, Phytoplankton, and Zooplankton Data and Analyses Gathered from Muskegon Lakes, 1972-1975.

Menzel, D.W. and N. Corwin. 1965. The measurement of total phosphorus in seawater based on the liberation of organically bound fractions by persulfate oxidation. Limnol. Oceanogr., 10:280-282.

Michigan Department of Natural Resources. 1976. Personal communication from Mr. George A. Jackson on June 4, 1976.

Michigan Water Resources Commission. 1967. White Lake Nutrient Survey. State of Michigan, Lansing. 46 pp.

Michigan Water Resources Commission. 1976. Industrial Wastewater Surveys, 1967-1975. Comprehensive Studies. State of Michigan, Lansing.

Mortimer, C.H. 1941. The exchange of dissolved substances between mud and water, I and II. J. Ecol., 29:280-329.

Mortimer, C.H. 1942. The exchange of dissolved substances between mud and water, III and IV. J. Ecol., 30:147-201.

Mortimer, C.H. 1971. Chemical exchanges between sediments and water in the Great Lakes—speculations on probable regulatory mechanisms. Limnol. Oceanogr., 16:387-404.

Murphy, J. and J.P. Riley. 1962. A modified single solution method for determination of phosphate in natural waters. Anal. Chim. Acta., 27:31-36.

Muskegon County. 1976. Muskegon County Wastewater Management System, Progress Report-Draft. Department of Public Works, Muskegon County, Michigan. 375 pp.

Neel, J.K., S.A. Peterson, and W.L. Smith. 1973. Weed Harvest and Lake Nutrient Dynamics. Office of Research and Monitoring, U.S. Environmental Protection Agency, Washington, D.C. 91 pp.

O'Brien, J.E. 1962. Automatic analysis of chlorides in sewage. Wastes Engineering, 33:670-672.

Porcella, D.B., J.S. Kumagai, and E.J. Middlebrooks. 1970. Biological effects on sediment-water nutrient interchange. J. San. Eng. Div., ASCE, 42:911-926.

Powers, C.F. and A. Robertson. 1965. Some Quantitative Aspects of the Macrobenthos of Lake Michigan. Proc. 8th Conf. of Great Lakes Res. pp. 153-159.

Powers, C.F. and A. Robertson. 1967. Design and Evaluation of an All-Purpose Benthos Sampler. The University of Michigan, Great Lakes Res. Div. Spec. Rept. No. 30:126-131.

Sands, H.D. 1974. A Study of the Biochemical Oxygen Demand, Chemical Oxygen Demand, and Total Organic Carbon Variations for the Muskegon Area. Unpublished manuscript.

Schelske, C.L. 1962. Iron, organic matter, and other factors limiting primary productivity in a marl lake. Science, 136:45-46.

Schelske, C.L. and E.F. Stoermer. 1971. Eutrophication, silica, and predicted changes in algal quality in Lake Michigan. Science, 173: 423-424.

Schwoerbel, J. 1970. Methods of Hydrobiology. Pergamon Press, New York, 200 pp.

Shapiro, J. and G. Glass. 1975. Synergistic effects of P and Mn on growth of Lake Superior algae. Verhand. Verein. Internat. Limnol., 19:395-404.

Strickland, J.D.H. and T.R. Parsons. 1968. A Pratical Handbook of Seawater Analysis. Fisheries Research Board of Canada, Ottawa. 311 pp.

Technicon Instrument Corporation. 1971. Chloride in Water and Wastewater, Industrial Method. Special Bulletin. Technicon, Inc., Tarrytown, New York. 2 pp.

Technicon Instrument Corporation. 1972. Nitrate and Nitrite in Water and Seawater, Industrial Method. Special Bulletin. Technicon, Inc., Tarrytown, New York. 2 pp.

Technicon Instrument Corporation. 1973a. Iron in Water and Seawater, Industrial Method. Special Bulletin. Technicon, Inc., Tarrytown, New York. 4pp.

Technicon Instrument Corporation. 1973b. Ammonia in Water and Seawater, Industrial Method. Special Bulletin. Technicon, Inc., Tarrytown, New York. 2 pp.

Technicon Instrument Corporation. 1973c. Ortho-Phosphate in Water and Seawater, Industrial Method. Special Bulletin. Technicon, Inc., Tarrytown, New York. 3 pp.

Technicon Instrument Corporation. 1973d. Silicates in Water and Seawater, Industrial Method. Special Bulletin. Technicon, Inc., Tarrytown, New York. 2 pp.

Thomas, W.H. 1970a. On nitrogen deficiency in tropical Pacific oceanic phytoplankton: photosynthesis parameters in poor and rich water. Limnol. Oceanogr., 15:380-385.

Thomas, W.H. 1970b. Effect of ammonium and nitrate concentration on chlorophyll increases in natural tropical Pacific phytoplankton populations. Limnol. Oceanogr., 15:386-394.

U.S. Environmental Protection Agency. 1974. Methods of Chemical Analysis of Water and Wastes. USEPA Office of Technology Transfer, Washington, D.C. 298 pp.

U.S. Environmental Protection Agency. 1975a. National Eutrophication Survey. White Lake, Muskegon County, Michigan. Working paper #217. Pacific Northwest Environmental Research Laboratory, Corvallis, Oregon. 42 pp.

U.S. Environmental Protection Agency. 1975b. A Compendium of Lake and Reservoir Data Collected by the National Eutrophication Survey in the Northeast and North Central United States. Working paper #474. Corvallis Environmental Research Laboratory, Corvallis, Oregon. 210 pp.

U.S. Environmental Protection Agency. 1975c. National Eutrophication Survey. Muskegon Lake, Muskegon County, Michigan. Working paper #203. Pacific Northwest Environmental Research Laboratory, Corvallis, Oregon, 38 pp.

U.S. Environmental Protection Agency. 1975d. National Eutrophication Survey. Mona Lake, Muskegon County, Michigan. Working paper #202. Pacific Northwest Environmental Research Laboratory, Corvallis, Oregon. 29 pp.

U.S. Geological Survey. 1971. Water Resources Data for Michigan. Part I—Surface Water Records. U.S. Dept. Interior, Washington, D.C. 271 pp.

U.S. Geological Survey. 1972. Water Resources Data for Michigan. Part I—Surface Water Records. U.S. Dept. Interior, Washington, D.C. 267 pp.

U.S. Geological Survey. 1973. Water Resources Data for Michigan. Part I—Surface Water Records. U.S. Dept. Interior, Washington, D.C. 270 pp.

U.S. Geological Survey. 1974. Water Resources Data for Michigan. Part I—Surface Water Records. U.S. Dept. Interior, Washington, D.C. 271 pp.

Vollenweider, R.A. 1969. Possibilities and limits of elementary models concerning the budget of substances in lakes. Arch. Hydrobiol., 66: 1-36.

Vollenweider, R.A. 1975. Input-output models with special reference to the phosphorus loading concept in limnology. Schweiz Z. Hydrol., 37: 53-83.

Weaver, G. 1976. Estimates of storm sewer contamination to Muskegon Lakes. Western Michigan Shoreline Federal 208 Commission. Personal communication.

Westlake, D.F. 1968. The Biology of Aquatic Plants and Their Management. Proc. 9th British Weed Control Conf., 372-381.

Wetzel, R.G. 1965. Necessity for decontamination of filters in C-14 measured rates of photosynthesis in fresh waters. Ecology, 46:540-542.

Zall, D.M., D. Fisher, and M.Q. Garner. 1956. Photometric determination of chlorides in water. Anal. Chem., 28:1665-1668.

APPENDIX A

MUSKEGON ALGAL NUTRIENT BIOASSAY STUDY

WHITE LAKE PROJECT

by

M. T. Auer and R. P. Canale

Department of Civil Engineering
The University of Michigan
Ann Arbor, Michigan 48105

for

Michigan Water Resources Commission
Department of Natural Resources
Lansing, Michigan 48926

EPA Grant No. G005104

SECTION 108 (a) PROGRAM
OFFICE OF GREAT LAKES COORDINATOR
U.S. ENVIRONMENTAL PROTECTION AGENCY, REGION V
CHICAGO, ILLINOIS 60604

ABSTRACT *

Algal nutrient bioassays were conducted on White Lake, Muskegon County, Michigan in November 1973, and April, July, and October 1974. Ammonia and nitrate nitrogen (both singly and combined), orthophosphate, and a trace metals mixture were added to large volumes of lake water possessing the natural phytoplankton assemblage. Growth, as measured by chlorophyll a concentration, was followed for a period of 24-28 days. Water chemistry and cell numbers were monitored throughout the course of the experiment.

Nutrient limitation in White Lake phytoplankton populations displayed a marked seasonality. Stimulation due to the addition of orthophosphate was observed to be most important in the fall. Nitrogen limitation was observed during the spring and summer.

The effect of supplying different forms of nitrogen is discussed. Observations and measurements of nitrification are presented and their importance is considered. A discussion of the interpretation of bioassay results is provided with reference to lake management objectives.

---

* The complete Appendix may be obtained from the National Technical Information Service, Springfield, VA. 22151.

APPENDIX B

SUBMERGED AQUATIC MACROPHYTES IN WHITE LAKE, MICHIGAN

by

P. L. Freedman and R. P. Canale

Department of Civil Engineering
The University of Michigan
Ann Arbor, Michigan 48105

for

Michigan Water Resources Commission
Department of Natural Resources
Lansing, Michigan 48926

EPA Grant No. G005104

SECTION 108 (a) PROGRAM
OFFICE OF GREAT LAKES COORDINATOR
U.S. ENVIRONMENTAL PROTECTION AGENCY, REGION V
CHICAGO, ILLINOIS 60604

ABSTRACT *

A field and laboratory study was conducted to investigate the importance of the submerged aquatic macrophytes to the White Lake ecosystem. Seven sampling cruises were conducted between June and October of 1974. The lake macrophyte community was mapped with respect to areal and biomass distribution. Samples of plants were later analyzed for carbon, hydrogen, nitrogen, potassium and phosphorus. The findings reveal that the areal extent of the macrophtes approaches one fifth of the lakes total surface area and the macrophyte biomass is equal in magnitude to that of the phytoplankton. The White Lake macrophyte community does not appear to be limited by phosphorus or nitrogen, and is probably restricted only by light and space requirements. The significance of the macrophyte nutrient content seems to be minor in comparison with other components of the lake nutrient content. The importance of macrophyte nutrients is, however, realized when compared with stream nutrient loads. If White Lake water quality improves as a consequence of the recently initiated wastewater diversion program, the macrophyte problem could become greater. Increased water clarity would increase the areal extent and possibly the density of the macrophyte community. The impact of the suspected sediment nutrient pumping by the macrophytes could also lessen or retard the lake response to the diversion.

---

* The complete Appendix may be obtained from the National Technical Information Service, Springfield, VA. 22151.

APPENDIX C

NUTRIENT RELEASE FROM ANAEROBIC SEDIMENTS IN WHITE LAKE, MICHIGAN

by

P. L. Freedman and R. P. Canale

Department of Civil Engineering
The University of Michigan
Ann Arbor, Michgian 48105

for

Michigan Water Resources Commission
Department of Natural Resources
Lansing, Michigan 48926

EPA Grant No. G005104

SECTION 108 (a) PROGRAM
OFFICE OF GREAT LAKES COORDINATOR
U.S. ENVIRONMENTAL PROTECTION AGENCY, REGION V
CHICAGO, ILLINOIS 60604

ABSTRACT *

A multiphase study was conducted to assess the significance of sediment nutrient release under anaerobic conditions in White Lake. This involved field monitoring, laboratory experiments, and *in situ* measurements. The results of these phases were in general agreement for both nitrogen and phosphorus. The average sediment nutrient release rates were 25.1 mg P/m^2/day soluble reactive phosphorus, 38.0 mg P/m^2/day total dissolved phosphorus and 36.2 mg N/m^2/day ammonia. The associated net diffusion coefficient across the sediment-water interface was calculated to be 1.0 to 1.5 x 10^{-5} cm^2/sec. Theoretical and experimental results for silicon release were not in good agreement. Speculative explanations for the discrepency were given.

Assessment of the influence of the sediments in lake water quality improvements suggest that the sediments can contribute nutrients to the overlying waters at levels sufficient to support troublesome plant growth. Consequently, improvements in water quality, resulting from the recent White Lake sewage diversion, could be delayed or reduced until after the lake sediments release the nutrients accumulated previously during the period of higher nutrient loading and until the sediments attain a new balance with the reduced loading conditions.

---

* The complete Appendix may be obtained from the National Technical Information Service, Springfield, VA. 22151.

APPENDIX D

MODEL PROJECTIONS OF PHOSPHORUS CONCENTRATIONS

by

R. P. Canale and W. S. Lung

Department of Civil Engineering
The University of Michigan
Ann Arbor, Michigan 48105

for

Michigan Water Resources Commission
Department of Natural Resources
Lansing, Michigan 48926

EPA Grant No. G005104

SECTION 108 (a) PROGRAM
OFFICE OF GREAT LAKES COORDINATOR
U.S. ENVIRONMENTAL PROTECTION AGENCY, REGION V
CHICAGO, ILLINOIS 60604

ABSTRACT *

Most previous phosphorus and euthrophication models have considered a single component (total phosphorus) and have been applied to lakes in which sediments play a minor role in regulating the phosphorus cycle. However, under many circumstances, other phenomena, such as phosphorus regeneration from the sediments and transformation of phosphorus within the sediment may complicate the response of the lake and require more comprehensive models.

A model which incorporates the water and the sediment systems and considers two forms of phosphorus (particulate and dissolved) is developed for White Lake, Michigan. Dynamic interactions between phosphorus in the sediments and the water are quantified by taking account of particulate phosphorus sinking to the sediment-water interface and diffusion of dissolved phosphorus across the interface. Other model mechanisms include vertical eddy diffusion in the water, phosphorus transformation between the particulate form and the dissolved form in both the water and the sediment, diffusion of phosphorus in the interstitial water, and sedimentation in the sediments.

Extensive field data have been used to determine the coefficients and parameters defined in the model formulations. Temperature data are used to estimate the vertical eddy diffusion coefficient. A non-linear biological production model and field data are used to calculate equivalent first-order kinetic coefficients in the water system. Literature is consulted to obtain the sinking velocity of particulate phosphorus in the lake. The diffusion coefficient in the interstitial water is determined by matching a calculated chloride profile with observed data in the sediment. The kinetic coefficients in the sediment are obtained by model tuning to match the observed phosphorus profiles in the sediment. The kinetic coefficient in the upper sediment layers correlates with the dissolved oxygen level of the bottom of the lake.

Close agreeement has been obtained between the model calculations and the observed data, especially for the upper layers of the sediment. Sensitivity analysis for the model further substantiates the model calculations. It is found that two separate forms of phosphorus are necessary to gain detailed insight into the dynamics of phosphorus cycling in White Lake. The model also explains massive releases of phosphorus from the sediment to the hypolimnion of White Lake in summer. It appears that the model may be applied to lakes having different degrees of eutrophication.

The model has been used to calculate long-term trends of phosphorus concentrations in White Lake. The results indicate some decreases of phosphorus levels in the first year after implementation of a nutrient reduction program and virtually no improvement after that. However,

the lake will probably not experience dramatic changes in phosphorus concentration because upstream non-point sources of phosphorus are important. Furthermore, year-to-year variations of these loadings overshadow the effect of nutrient reduction. Therefore, additional schemes of the lake rehabilitation are evaluated using the model in order to achieve further decreases of phosphorus concentrations in White Lake. It is found that further reduction of the present non-point phosphorus loading is necessary for full recovery, although significant additional improvements are also expected from aeration and mixing.

---

* The complete Appendix may be obtained from the National Technical Information Service, Springfield, VA. 22151.

APPENDIX E

WATER QUALITY AND LOADING RATE DATA FOR THE MUSKEGON COUNTY TREATMENT SYSTEM

The following ten tables summarize water quality and loading rate data for the Muskegon and Whitehall Spray-Irrigation Treatment Sites.

Table 1 lists the average annual reduction in concentrates of selected parameters between the treatment site influent and spray drainage affluent at Muskegon spray site. Table 2 defines the associated annual average flows and loads. Tables 3 through 8 give similar data expressed on a monthly basis. Tables 9 and 10 contain average annual concentration and estimated flow and loading data for the Whitehall Spray Treatment System.

Table 1. AVERAGE ANNUAL CONCENTRATIONS OF SELECTED POLLUTANTS IN INFLUENT AND EFFLUENT AT THE MUSKEGON TREATMENT SITE

	PO_4-P	TP	NH_4-N	(NO_2+NO_3)-N	TKN	Cl	Fe	SO_4
				- mg/ℓ -				
Influent								
1973*	1.28	1.79	3.58	0.03	6.30	154	0.94	80.0
1974	1.83	2.70	7.91	0.07	10.60	176	1.02	82.0
1975	1.56	2.38	6.12	--	8.24	182	0.79	75.0
Effluent, Mosquito Creek								
1973*	0.03	--	0.35	0.18	--	57.0	2.13	78.0
1974	0.03	--	0.48	1.20	--	60.0	1.68	83.0
1975	0.03	--	0.59	2.09	--	78.0	1.03	81.0
Effluent, Big Black Creek								
1973*	0.02	--	0.34	0.15	--	21.0	14.6	214
1974	0.02	--	0.50	1.04	--	16.0	22.0	327
1975	0.02	--	0.48	0.95	--	32.0	17.4	284

*Six-month period (July-December).

Table 2. ANNUAL WASTEWATER FLOWS AND LOADS OF NITROGEN AND PHOSPHORUS INFLUENT AND EFFLUENT AT THE MUSKEGON TREATMENT SITE

	Nutrient	Year		
		1973*	1974	1975
Influent				
Flow, TCM/year		18,798	37,755	36,529
Load, kgms/year	PO_4-P	24,061	69,091	56,985
	TP**	33,648	101,938	86,939
	NH_4-N	67,297	298,642	223,557
	TKN**	118,427	400,203	300,999
Effluent, Black Creek				
Flow, TCM/year		5,494	9,494	13,385
Load, kgms/year	PO_4-P	110	190	268
	TP***	160	277	391
	$(NH_4 + NO_2 + NO_3)$-N	2,692	14,620	19,141
Effluent, Mosquito Creek				
Flow, TCM/year		11,443	35,105	35,193
Load, kgms/year	PO_4-P	343	1,053	1,056
	TP***	500	1,537	1,541
	$(NH_4 + NO_2 + NO_3)$-N	6,065	58,976	94,317

* TCM flow for six-month period (July-December in 1973).

** Limited data, therefore missing values estimated by linear regression with PO_4-P and NH_4 estimators for TP and TKN.

*** Based on a 1.46 TP:PO_4 ratio obtained from plant influent data.

Table 3. MONTHLY INFLUENT CONCENTRATIONS OF PHOSPHORUS AND NITROGEN TO MUSKEGON WASTEWATER SYSTEM

Month/Year	PO_4-P mg/ℓ	NH_4 mg/ℓ
7/73	1.02	3.39
8/73	0.91	3.33
9/73	2.61	3.87
10/73	1.15	3.72
11/73	0.85	3.03
12/73	1.28	4.15
MEAN 1973	1.30	3.58
1/74	1.12	3.95
2/74	1.60	5.96
3/74	1.52	4.63
4/74	1.88	7.08
5/74	1.70	9.06
6/74	1.91	7.15
7/74	2.16	10.70
8/74	2.15	8.30
9/74	2.33	10.96
10/74	2.13	9.19
11/74	2.19	8.61
12/74	1.87	9.32
MEAN 1974	1.88	7.91
1/75	1.86	9.29
2/75	1.83	8.56
3/75	1.89	12.90
4/75	1.41	7.80
5/75	1.74	8.40
6/75	1.66	7.25
7/75	1.50	5.85
8/75	1.23	3.27
9/75	1.05	2.27
10/75	0.96	1.56
11/75	1.72	2.94
12/75	1.86	3.34
MEAN 1975	1.56	6.12

Table 4. MONTHLY INFLUENT CONCENTRATIONS OF PHOSPHORUS AND NITROGEN TO MUSKEGON WASTEWATER SYSTEM

Month/Year	Flow TCM	PO_4-P kg	NH_4 kg
7/73	3,278	3,344	11,112
8/73	3,353	3,051	11,165
9/73	3,133	8,177	12,125
10/73	3,176	3,652	11,815
11/73	2,965	2,520	8,984
12/73	2,893	3,703	12,006
TOTAL 1973	18,798	24,447	67,207
1/74	3,155	3,534	12,462
2/74	2,808	4,493	16,736
3/74	3,277	4,981	15,173
4/74	3,198	6,012	22,642
5/74	3,301	5,612	29,907
6/74	3,411	6,515	24,389
7/74	3,115	6,728	33,331
8/74	3,453	7,424	28,660
9/74	3,014	7,023	33,033
10/74	3,145	6,699	28,903
11/74	2,087	4,571	17,969
12/74	2,791	5,219	26,012
TOTAL 1974	37,755	68,811	289,217
1/75	2,984	5,550	27,721
2/75	2,659	4,866	22,761
3/75	2,779	5,252	35,849
4/75	2,928	4,128	22,838
5/75	3,188	5,547	26,779
6/75	3,245	5,387	23,526
7/75	2,872	4,308	16,801
8/75	3,190	3,924	10,431
9/75	3,135	3,292	7,241
10/75	3,380	3,245	5,273
11/75	3,083	5,303	9,064
12/75	3,086	5,740	10,307
TOTAL 1975	36,529	56,542	218,591

Table 5. MONTHLY CONCENTRATIONS OF PHOSPHORUS AND NITROGEN IN EFFLUENT AT MOSQUITO CREEK OUTFALL

Month/Year	$(NH_4 + NO_2 + NO_3)$-N mg/ℓ	PO_4-P mg/ℓ
7/73	0.31	0.033
8/73	0.41	0.027
9/73	0.40	0.023
10/73	0.57	0.060
11/73	0.61	0.034
12/73	0.89	--
MEAN 1973	0.53	0.029
1/74	0.81	--
2/74	1.04	--
3/74	0.99	0.064
4/74	1.21	0.064
5/74	1.32	0.032
6/74	1.50	0.047
7/74	2.23	0.049
8/74	2.34	0.023
9/74	2.20	0.006
10/74	2.59	0.013
11/74	2.50	0.026
12/74	1.42	0.075
MEAN 1974	1.68	0.033
		est.
1/75	2.50	0.03
2/75	2.50	0.03
3/75	2.08	0.03
4/75	1.99	0.03
5/75	2.50	0.03
6/75	2.35	0.03
7/75	3.41	0.03
8/75	3.49	0.03
9/75	3.12	0.03
10/75	2.93	0.03
11/75	2.94	0.03
12/75	2.37	0.03
MEAN 1975	2.68	0.03

Table 6. MONTHLY EFFLUENT FLOW AND LOADS OF PHOSPHORUS AND NITROGEN TO MOSQUITO CREEK FROM THE MUSKEGON TREATMENT SITE

Month/Year	Flow TCM	$(NH_4 + NO_2 + NO_3)$-N kg	PO_4-P kg
7/73	1,927	597	63.6
8/73	1,926	790	52.0
9/73	1,869	748	43.0
10/73	1,926	1,098	115.6
11/73	1,869	1,140	63.6
12/73	1,926	1,714	--
TOTAL 1973	11,443	6,087	338.8
1/74	1,476	1,195	--
2/74	1,333	1,387	--
3/74	1,828	1,810	117.0
4/74	2,337	2,828	149.6
5/74	2,180	2,878	69.8
6/74	3,586	5,379	168.5
7/74	5,348	11,927	262.1
8/74	4,292	10,044	98.7
9/74	4,040	8,888	24.2
10/74	3,471	8,990	45.1
11/74	2,564	6,410	66.7
12/74	2,650	3,762	198.8
TOTAL 1974	35,105	65,498	1,200.5
1/75	3,599	8,998	108.0
2/75	3,252	8.127	97.5
3/75	1,956	4,069	58.7
4/75	1,893	3,768	56.8
5/75	1,956	4,891	58.7
6/75	2,461	5,784	73.8
7/75	2,895	9,872	86.8
8/75	2,426	8,466	73.1
9/75	4,846	16,087	145.4
10/75	2,191	6,420	65.7
11/75	5,527	16,249	165.8
12/75	2,191	5,193	65.7
TOTAL 1975	35,193	97,924	1,056.0

Table 7. MONTHLY CONCENTRATIONS OF PHOSPHORUS AND NITROGEN IN EFFLUENT AT BLACK CREEK OUTFALL

Month/Year	PO_4-P mg/ℓ	$(NH_4 + NO_2 + NO_3)$-N mg/ℓ
7/73	0.03	0.41
8/73	--	0.37
9/73	0.01	0.31
10/73	0.04	0.45
11/73	--	0.45
12/75	0.00	0.97
MEAN 1973	0.02	0.49
1/74	--	0.98
2/74	--	1.15
3/74	--	1.08
4/74	--	1.53
5/74	--	1.68
6/74	--	1.62
7/74	0.04	1.43
8/74	0.01	2.04
9/74	0.04	1.52
10/74	0.02	2.17
11/74	0.02	1.91
12/74	0.02	1.40
MEAN 1974	0.02	1.54
1/75	0.03	1.48
2/75	0.04	1.40
3/75	0.01	1.42
4/75	0.01	1.45
5/75	--	1.56
6/75	0.01	1.45
7/75	0.01	1.81
8/75	0.03	1.60
9/75	0.01	1.67
10/75	0.01	1.27
11/75	0.03	0.95
12/75	0.01	1.14
MEAN 1975	0.02	1.43

Table 8. MONTHLY EFFLUENT FLOW AND LOADS OF PHOSPHORUS AND NITROGEN TO BLACK CREEK FROM MUSKEGON TREATMENT SITE

Month/Year	Flow TCM	PO_4-P kg	$(NH_4 + NO_2 + NO_3)$-N kg
7/73	1,408	42.2	577
8/73	1,604	--	593
9/73	1,348	13.5	418
10/73	516	20.6	232
11/73	276	--	124
12/73	342	--	332
TOTAL 1973	5,494	76.3	2,276
1/74	475	--	466
2/74	535	--	615
3/74	625	--	675
4/74	864	--	1,322
5/74	1,096	--	1,841
6/74	676	--	1,095
7/74	758	30.3	1,084
8/74	1,885	18.8	3,845
9/74	782	31.3	1,189
10/74	504	10.0	1,094
11/74	874	17.5	1,669
12/74	420	8.4	588
TOTAL 1974	9,494	116.4	15,483
1/75	601	18.0	889
2/75	925	37.0	1,295
3/75	1,047	10.5	1,487
4/75	980	9.8	1,421
5/75	940	--	1,466
6/75	1,153	11.5	1,672
7/75	1,277	12.8	2,311
8/75	1,181	35.4	1,877
9/75	1,801	18.0	3,008
10/75	1,232	12.3	1,565
11/75	1,048	31.4	996
12/75	1,200	12.0	1,368
TOTAL 1975	13,385	209.7	19,365

Table 9. AVERAGE ANNUAL CONCENTRATIONS OF SELECTED POLLUTANTS IN INFLUENT AND EFFLUENT (WELLS) AT THE WHITEHALL TREATMENT SITE

	PO_4-P	TP	NH_4-N	$(NO_2 + NO_3)$-N	TKN	Cl	SO_4
	- mg/ℓ -						
Influent							
1974	--	--	10.46	--	--	--	--
1975	0.35	0.65	38.22	0.67	48.27	--	--
Effluent, (wells)*							
1974	0.04	--	0.20	0.07	--	5.4	18.3
1975	0.02	0.05	0.05	0.06	--	3.3	44.5

*Average of parameter concentration in perimeter wells at Whitehall.

Table 10. ANNUAL WASTEWATER FLOWS AND LOADS OF NITROGEN AND PHOSPHORUS, INFLUENT AND EFFLUENT AT THE WHITEHALL TREATMENT SITE

	Nutrient	Year 1974	1975
Influent			
Flow, TCM/year		733	1,505
Load, kgms/year	PO_4-P	--	533
	TP	--	978
	NH_4-N	7,667	57,521
	$(NO_2 + NO_3)$-N	--	1,007
	TKN	--	72,646
Effluent (wells)*			
Flow, TCM/year		6.32	298.41
Load, kgms/year	PO_4-P	0.25	5.97
	TP	--	14.92
	NH_4	1.26	14.92
	$(NO_2 + NO_3)$-N	0.44	17.90

APPENDIX F

INDUSTRIAL AND MUNICIPAL DISCHARGE INVENTORY

Tables 1, 2, and 3 list the industrial discharge for Muskegon, Mona, and White Lake basins. The industries in these tables are restricted to those which have filed National Permit Discharge Elimination System (NPDES) reports with the State of Michigan Environmental Protection Bureau. Included in these tables is documentation of the industry, NPDES discharge number, discharge type and character, and waste flow. Table 4 documents the status of various Muskegon County communities with respect to their participation in the Muskegon County wastewater treatment system. In all tables industries and municipalities have been categorized into those with more than 50% and less than 50% correction to the wastewater treatment system.

Table 1. INDUSTRIAL NPDES WASTEWATER DISCHARGES TO MUSKEGON LAKE

Industry	Year	State Outflow Number	Discharge Type*	% Processing	% Cooling	% Sanitary	Flow (Mgd)
Industries with more than 50% connected to the Spray-Irrigation Treatment System							
Amstead Industries Inc. Standard Automotive Parts	1975	610185	6	6	94	0	0.0005
		610186	6	0	100	0	0.0240
		610234	6	99	1	0	0.0006
		610235	2	100	0	0	0.0004
		980537	8	0	45	55	0.0290
	1974	610185	6	100	0	0	0.0030
		610186	6	0	100	0	0.0060
		610234	6	95	5	0	0.0040
		610235	2	100	0	0	0.0009
		980537	6	0	43	57	0.0510
	1973	971832	2	0	1	99	0.0020
		971833	2	100	0	0	0.0020
		971834	2	0	100	0	0.0040
		971835	2	100	0	0	0.0005
Anaconda Wire and Cable Co. Division of Anaconda Co.	1975	610156	8	5	82	13	0.1152
	1974	610156	8	5	82	13	0.1459
	1973	619156	8	5	82	13	0.0136
Breneman, Inc. Muskegon Division	1974	610158	6	100	0	0	0.0001
		970042	8	46	19	35	0.0245
	1973	610158	8	46	19	35	0.0243
		970042	6	100	0	0	0.0001
Great Lakes Plating Corp.	1975	610165	8	75	20	5	0.025
	1974	610165	8	80	10	10	0.050
	1973	610165	8	82	10	8	0.018
Michigan Celery Promotion Co-op, Inc.	1975	610233	8	65	25	5	0.0246
	1973	970730	8	80	20	0	0.071
		970731	4	0	0	100	0.001
Michigan Spring Co.	1975	940222	8	0	100	0	0.002
	1974	940223	8	100	0	0	0.009
	1973	970716	8	100	0	0	0.008
		970717	8	0	100	0	0.002
Muskegon Piston Ring Co. Muskegon Plant	1975	610179	1	0	100	0	0.0002
		940240	8	35	55	10	0.5222
	1974	610179	1	0	100	0	0.064
	1973	610179	1	0	100	0	0.064
		970609	8	98	0	2	0.470
S.D. Warren Co.	1974	610232	1	0	100	0	4.1
		980451	8	0	0	100	0.14
		980452	8	100	0	0	15.4
	1973	610071	1	100	0	0	4.2
		610072	1	70	30	0	11.67
		970346	1	100	0	0	0.045
		970347	1	100	0	0	0.35
		970348	1	100	0	0	0.18
		970349	8	0	0	100	0.14
		970350	8	100	0	0	14.0
Standard Oil Car Wash	1975	880970	8	95	0	5	0.0118

Table 1.--continued

Industry	Year	State Outflow Number	Discharge Type*	% Processing	% Cooling	% Sanitary	Flow (Mgd)
Storey Chemical Co. Ott Division	1975	610085	2	0	100	0	0.55
		980563	8	68	30	2	0.75
	1974	610085	2	0	100	0	0.7
		980563	8	90	60	0	0.8
	1973	610084	1	60	40	0	0.75
		610085	2	0	100	0	0.55
Teledyne Continental Motors	1975	610063	1	0	100	0	0.25
		980785	8	0	0	100	0.05
		980786	8	100	0	0	0.70
	1974	610063	1	0	100	0	0.25
		980785	8	0	0	100	0.1
		980786	8	100	0	0	0.6
	1973	970849	1	100	0	0	0.77
		970850	1	100	0	0	0.77
Shaw Walker Co.	1975	980280	8	28	45	27	0.06
		980282	8	0	0	100	0.007
		980283	8	0	0	100	0.004
		980284	8	0	0	100	0.0004
	1975	980285	8	0	0	100	0.0007
		980286	8	90	0	10	0.0110
		980287	8	90	0	10	0.0680
		980288	8	93	57	0	0.0490
	1974	980280	8	25	47	28	0.056
		980282	8	25	47	28	0.100
		980283	8	0	0	100	0.005
		980284	8	0	0	100	0.0004
		980285	8	0	0	100	0.0009
		980286	8	87	2	11	0.0720
		980287	8	90	0	10	0.0096
	1973	970128	8	10	72	18	0.1548
		970129	8	0	0	100	0.0127
		970130	8	0	0	100	0.0060
		970131	8	0	0	100	0.0003
		970132	8	0	0	100	0.0009
		970133	8	88	12	0	0.0715
		970134	8	90	0	10	0.0105
Industries with less than 50% connected to the Muskegon Wastewater Treament System							
Campbell, Wyatt & Cannon	1975	610125	1	1	100	0	0.79
	1974	610125	1	0	100	0	0.5
	1973	610125	1	10	90	0	0.58
Cannon-Muskegon Corp.	1975	610214	2	0	99	1	0.20
	1974	610214	2	0	99	1	0.123
	1973	610214	2	0	99	1	0.123
Clark Gravely Corp.	1975	610213	1	0	100	0	0.0024
	1974	610213	1	0	100	0	0.0024
	1973	610213	1	0	100	0	0.0057
Geerpres Winger, Inc.	1975	610224	4	7	80	13	0.0078
	1974	610224	4	7	80	13	0.0078
	1973	610224	4	7	80	13	0.0078

Table 1.--continued

Industry	Year	State Outflow Number	Discharge Type*	% Processing	% Cooling	% Sanitary	Flow (Mgd)
Enterprise Brassworks	1975	610216	1	0	100	0	0.003
	1974	610216	1	0	100	0	0.014
	1973	610216	1	0	100	0	0.0204
Howmett Corp. Misco Division Plant #2	1975	Inactive					
	1974	610154	1	10	90	0	0.2
		610231	1	20	80	0	0.3
	1973	610154	1	20	80	0	0.3
		610231	1	10	90	0	0.24
Keene Corp., Kaydon Bearing	1975	610101	1	0	100	0	0.06
		610239	2	100	0	0	0.04
	1974	610101	1	0	100	0	0.06
		610239	2	100	0	0	0.04
	1973	610101	1	0	100	0	0.11
		610239	2	100	0	0	0.001
		970446	2	100	0	0	0.008
		970447	8	0	0	100	0.0039
Michigan Foundry Supply Co.	1975	660099	2	15	84	1	0.056
	1974	610099	2	15	84	1	0.056
	1973	610099	1	15	85	0	0.056
Naph-Sol Refinery Co.	1975	610123	1	100	0	0	0.066
	1974	610123	1	100	0	0	0.005
Port City Inc.	1975	610219	1	0	100	0	0.0004
		970213	8	6	0	94	0.0001
	1974	610219	1	0	100	0	0.0004
		980208	8	6	0	94	0.0001
	1973	610219	1	0	100	0	0.0004
		970213	8	0	6	94	0.0001
Sealed Power Co. Stanford Street	1975	610261	1	28	72	0	0.97
		980779	8	0	0	100	0.02
		980780	8	6	28	66	0.059
	1974	610261	1	28	72	0	1.034
		980779	8	0	0	100	0.0575
		980780	8	6	28	66	0.0575
	1973	971267	1	28	72	0	1.142
Seal Tex Co.	1974	927883	1	0	100	0	0.001
	1973	927883	1	0	100	0	0.001
Teledyne Continental Motors, N. Getty	1975	610076	1	0	100	0	0.05
		610102	2	100	0	0	0.028
		980927	8	0	0	100	0.035
	1974	610076	1	0	100	0	0.30
		610102	2	100	0	0	0.028
		980927	8	0	0	100	0.03
	1973	610076	1	0	100	0	0.26
		610102	2	100	0	0	0.028
		970797	8	0	100	0	0.03
West Michigan Dock & Market Corp.	1975	610418	1	0	100	0	0.154
	1974	610918	1	0	100	0	0.2
	1973	610418	1	0	100	0	0.2
Western Corp.	1975	610255	1	0	100	0	0.5
	1974	610255	1	0	100	0	0.5
	1973	610222	1	0	100	0	0.452
		610223	1	0	100	0	0.013
		970157	8	0	89	11	0.078

*The State of Michigan Environmental Protection Bureau categorizes the discharge types as follows: 1) surface; 2) lagoon; 3) spray; 4) septic tile field; 5) deep well; 6) surface overground flow; 7) other; 8) municipal (sanitary sewer).

Table 2. INDUSTRIAL NPDES WASTEWATER DISCHARGES TO MONA LAKE

Industry	Year	State Outflow Number	Discharge Type*	% Processing	% Cooling	% Sanitary	Flow (Mgd)
Industries with more than 50% connected to the Muskegon Wastewater Treatment System							
American Coil Spring Co.	1975	610246	1	30	52	18	0.115
		980092	8	70	5	25	0.120
	1974	610150	1	30	52	18	0.09
		980092	8	70	5	25	0.02
	1973	610150	1	5	70	25	0.103
		970023	8	70	5	25	0.038
Bennett Pump Co.	1975	610157	8	1	80	19	0.07
	1973	610157	8	1	80	19	0.06
Brown-Morse Co. Plant #1 110 E. Broadway	1973	610159	1	21	79	0	0.0082
		970440	8	28	6	66	0.0159
Brown-Morse Co. 6th Street	1975	610242	1	27	73	0	0.0056
		980277	8	28	6	66	0.0165
	1974	610242	1	27	73	0	0.0089
		980277	8	28	6	66	0.0165
	1973	610242	1	27	73	0	0.0089
		970441	8	28	6	66	0.0106
Brunswick Corp.	1975	610168	8	35	20	45	0.27
	1974	610168	8	46	21	33	0.42
	1973	610168	8	46	21	33	0.42
Burdick & Jackson Laboratories	1975	610160	8	5	92	3	0.0629
	1974	610160	8	5	92	3	0.0336
	1973	610160	8	5	92	3	0.0290
Johnson Products Division Sealed Power Corp.	1975	610166	1	0	100	0	0.034
		980447	8	71	12	17	0.256
	1974	610166	1	0	100	0	0.115
		980447	8	40	43	17	0.173
	1973	610166	1	60	40	0	0.076
		970742	8	24	26	10	0.114
Tek Mold Inc.	1975	610208	8	10	86	4	0.03
	1974	610208	8	10	86	4	0.1387
	1973	610208	8	10	86	4	0.1864
Universal Camshaft Co.	1975	924374	8	35	60	5	0.0335
		980441	7	100	0	0	0.0006
	1974	924374	8	35	60	5	0.8335
		980441	4	100	0	0	0.0001
	1973	924374	8	35	60	5	0.0335
		970659	4	100	0	0	0.0001
Industries with less than 50% connected to Muskegon Wastewater Treatment System							
American Porcelain Enamel	1975	610155	1	100	0	0	0.612
		980455	8	0	0	100	0.0026
	1974	610155	1	100	0	0	0.0538
		980455	8	0	0	100	0.0018
	1973	610155	1	100	0	0	0.0564
		970251	8	0	0	100	0.0011

Table 2.--continued

Industry	Year	State Outflow Number	Discharge Type*	% Processing	% Cooling	% Sanitary	Flow (Mgd)
Campbell, Wyatt & Cannon Plant #1	1975	610073	1	20	80	0	0.28
	1974	610073	1	20	80	0	0.28
	1973	610073	1	5	95	0	0.28
Campbell, Wyatt & Cannon Plant #4	1975	610126	1	0	100	0	0.08
	1974	610126	1	0	100	0	0.08
	1973	610126	1	1	99	0	0.11
Coil Anodizers Division of Lorin Industries (Kersman Co.)	1975	610057	1	50	50	0	0.250
		610059	2	100	0	0	0.002
		980008	8	0	0	100	0.001
	1974	610057	1	50	50	0	0.400
		610059	2	100	0	0	0.002
		980008	8	0	0	100	0.0001
	1973	610057	1	50	50	0	0.500
		610059	2	100	0	0	0.011
		970300	2	0	100	0	0.144
East Shore Chemical Co.	1975	610162	1	0	100	0	0.029
		940264	8	95	0	5	0.006
	1974	610162	1	0	100	0	0.08
		980877	8	95	0	5	0.005
	1973	610162	1	0	100	0	0.05
		970475	8	95	0	5	0.005
Fleet Engineers, Inc.	1975	610241	2	100	0	0	0.0016
		940245	8	0	0	100	0.0008
	1974	610241	2	100	0	0	0.0001
	1973	970456	2	100	0	0	0.0001
		970457	8	0	0	100	0.0008
Hazekomp Bert & Son	1975	610119	2	11	88	1	0.01
	1974	610119	2	11	88	1	0.01
Lakeway Chemicals, Inc.	1975	610095	2	13	87	0	1.3
		980751	4	0	0	100	0.002
	1974	610095	2	13	87	0	0.76
		980757	4	0	0	100	0.002
	1973	610095	2	13	87	0	0.78
		970029	4	0	0	100	0.002
Peerless Plating Co.	1975	610200	2	49	51	0	0.012
		980073	4	0	0	100	0.0001
	1974	610200	2	33	67	0	0.0082
		980073	4	0	0	100	0.0001
	1973	610200	2	42	58	0	0.0165
		970138	4	0	0	100	0.0000
Punches & Son Slaughter House	1975	610120	2	50	40	10	0.0005
	1973	610120	2	25	25	50	0.001
Ryerson Creek Produce Inc.	1975	610218	1	99	1	0	0.0010
	1973	610218	1	98	1	1	0.0005
Sealed Power Corp. Harvey St.	1975	610248	1	0	100	0	?
	1974	610170	1	0	100	0	0.0504
		980777	8	0	0	100	0.0013
		980778	8	36	57	7	0.0618
	1973	971270	1	5	95	0	0.055

Table 2.--continued

Industry	Year	State Outflow Number	Discharge Type*	% Processing	% Cooling	% Sanitary	Flow (Mgd)
Stresser Joseph Laundry	1975	610053	2	0	0	100	0.0028
	1974	610053	2	100	0	0	0.001
	1973	610053	2	0	0	100	0.002
Thomas Solvent Co. Thermal Chemical Inc.	1975	610078	2	1	98	1	0.0045
	1974	610078	2	1	98	1	0.0045
	1973	610078	2	1	98	1	0.0045
Unico Inc.	1975	610228	2	100	0	0	0.0015
		970012	8	0	0	100	0.0001
	1974	610228	2	100	0	0	0.0021
	1973	610228	2	100	0	0	0.0021
		970012	8	0	0	100	0.0002

*The State of Michigan Environmental Protection Bureau categorizes the discharge types as follows: 1) surface; 2) lagoon; 3) spray; 4) septic tile field; 5) deep wall; 6) surface overground flow; 7) other; 8) municipal (sanitary sewer).

Table 3. INDUSTRIAL NPDES WASTEWATER DISCHARGES TO WHITE LAKE

Industry	Year	State Outflow Number	Discharge Type*	% Processing	% Cooling	% Sanitary	Flow (Mgd)
Industries with more than 50% connected to Whitehall Wastewater Treatment System							
Whitehall Leather Co. Division Genesco Industries	1975	610047	8	85	4	1	0.4960
	1974	610062	1	85	15	0	0.5
		980565	4	0	0	100	0.004
	1973	610062	1	85	15	0	0.491
		970706	4	0	0	100	0.004
Industries with less than 50% connected to Whitehall Wastewater Treatment System							
Certified Concerete Inc.	1975	924137	2	100	0	0	0.0005
	1974	924137	2	100	0	0	0.0005
Happs Wash King	1974	610049	2	0	0	100	0.0005
	1973	610049	2	100	0	0	0.0005
Hooker Chemical Corp.	1975	610094		1	98	1	11.7
	1974	610094	1	1	98	1	12.2
	1973	610094	1	1	98	1	14.1
Howmett Corp., Misco Div. Plant #1	1973	610091	1	20	80	0	0.02
Howmett Corp., Misco Div. Plant #3	1975	610092	1	20	80	0	0.10
	1974	610092	1	20	80	0	0.104
	1973	610093	1	20	80	0	0.092
Howmett Corp., Misco Div. Plant #4	1975	610217	2	10	90	0	0.200
		980357	8	30	10	60	0.022
	1974	610217	2	10	90	0	0.33
		980357	8	30	10	60	0.036
	1973	610217					
		980357	2	10	90	0	0.033
Pin Key Manufacturing Co.	1975	927739	2	100	0	0	0.0002
		980640	4	0	0	100	0.0002
	1974	927739	2	0	0	100	0.0002
		980640	4	100	0	0	0.0002
	1973	927739	2	100	0	0	0.0002
		970786	4	0	0	100	0.0002
Tech Cast Inc.	1975	927742	2	0	100	0	0.02
	1974	927742	2	0	100	0	0.04

*The State of Michigan Environmental Protection Bureau categorizes the discharge types as follows: 1) surface; 2) lagoon; 3) spray; 4) septic tile field; 5) deep well; 6) surface overground flow; 7) other; 8) municipal (sanitary sewer).

Table 4. STATUS OF MUSKEGON COUNTY MUNICIPALITIES PARTICIPATION IN THE SPRAY-IRRIGATION WASTEWATER TREATMENT SYSTEM

Muskegon Lake Municipalities

Communities with more than 50% connected to the Muskegon Wastewater System

1. Muskegon
2. North Muskegon

Communities with less than 50% connected to the Muskegon Wastewater System

1. Dalton Township
2. Egelton Township
3. Laketon Township
4. Muskegon Township

Mona Lake Municipalities

Communities with more than 50% connected to the Muskegon Wastewater System

1. Muskegon Heights
2. Roosevelt Park

Communities with less than 50% connected to the Muskegon Wastewater System

1. Egelton Township
2. Fruitport Township
3. Muskegon Township
4. Norton Shores
5. Sullivan Township

White Lake Municipalities

Communities with more than 50% connected to the Whitehall Wastewater System

1. Whitehall

Communities with less than 50% connected to the Whitehall Wastewater System

1. Fruitland Township
2. Lakewood Club
3. Montague
4. Montague Township
5. Whitehall Township
6. White River Township

APPENDIX G

AVAILABILITY OF DATA FROM U.S. EPA STORAGE AND RETRIEVAL COMPUTER SYSTEM "STORET"

Much of the data collected has been entered into the U.S. Environmental Protection Agency's storage and retrieval computer system called "STORET". Access to STORET and use of this data may be obtained directly by authorized users or indirectly through any of the ten different EPA Regional Offices or STORET User Assistance, U.S. EPA, 401 M Street, S.W., Washington, D. C. 20460, phone (202) 426-7792. The applicable codes for gaining access to this data are: Agency Code No. 21 MI MUSK; State Code No. 26; County Code No. 121.